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Ecological Risks of Shale Oil and Gas Development to Wildlife, Aquatic Resources and their Habitats

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ABSTRACT: Technological advances in hydraulic fracturing and horizontal drilling have led to the exploration and exploitation of shale oil and gas both nationally and internationally. Extensive development of shale resources has occurred within the United States over the past decade, yet full build out is not expected to occur for years. Moreover, countries across the globe have large shale resources and are beginning to explore extraction of these resources. Extraction of shale resources is a multistep process that includes site identification, well pad and infrastructure development, well drilling, high-volume hydraulic fracturing and production; each with its own propensity to affect associated ecosystems. Some potential effects, for example from well pad, road and pipeline development, will likely be similar to other anthropogenic activities like conventional gas drilling, land clearing, exurban and agricultural development and surface mining (e.g., habitat fragmentation and sedimentation). Therefore, we can use the large body of literature available on the ecological effects of these activities to estimate potential effects from shale development on nearby ecosystems. However, other effects, such as accidental release of wastewaters, are novel to the shale gas extraction process making it harder to predict potential outcomes. Here, we review current knowledge of the effects of high-volume hydraulic fracturing coupled with horizontal drilling on terrestrial and aquatic ecosystems in the contiguous United States, an area that includes 20 shale plays many of which have experienced extensive development over the past decade. We conclude that species and habitats most at risk are ones where there is an extensive overlap between a species range or habitat type and one of the shale plays (leading to high vulnerability) coupled with intrinsic characteristics such as limited range, small population size, specialized habitat requirements, and high sensitivity to disturbance. Examples include core forest habitat and forest specialists, sagebrush habitat and specialists, vernal pond inhabitants and stream biota. We suggest five general areas of research and monitoring that could aid in development of effective guidelines and policies to minimize negative impacts and protect vulnerable species and ecosystems: (1) spatial analyses, (2) species-based modeling, (3) vulnerability assessments, (4) ecoregional assessments, and (5) threshold and toxicity evaluations.

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■ INTRODUCTION

Domestic energy production is a national priority and unconventional oil and gas resources are an important and growing source of energy globally. In its 2013 shale oil and gas assessment, the U.S. Energy Information Administration (USEIA) estimated 7299 trillion cubic feet (TCF) of technically recoverable shale gas and 345 billion barrels of technically recoverable shale oil in 41 countries, including the United States.[1] Countries included in the report span the globe, suggesting the potential footprint of developing these resources will encapsulate large portions of the earth. The United States is currently the leading nation in developing unconventional oil and gas resources and over the past decade has greatly increased its production efforts from shale, yet has only 9.1% of the EIA estimated total recoverable shale gas worldwide (665 TCF).[1] Documenting potential effects of such development to associated ecosystems during the current unconventional oil and gas boom will help guide future development throughout the world and inform adaptive management related to current activities.

In the continental United States, there are 20 shale plays (covering over 766 636 km^2),[2] and development of these shale resources is an increasingly important driver of landscape change. The shale layers are found in a number of regions across the contiguous U.S. and underlie a variety of landcover types (Figure 1) which include ecosystems with high biodiversity and regional and global significance including the Eastern deciduous forest, shrubland and sagebrush-steppe, grasslands and other herbaceous habitats.[3−5] In addition, there are numerous streams, rivers, wetlands, and other aquatic habitats potentially affected by development.

Extraction of oil and gas from shale resources is a multistep process that includes site identification, well pad and infrastructure construction, well drilling, high-volume hydraulic fracturing (HVHF), and then production which may include additional HVHF. High-volume hydraulic fracturing is used to fracture the shale and release the gas. While the technique has been used in some form for decades, the combination of HVHF with advanced horizontal drilling technologies has opened up large, previously uneconomical reserves. The process occurs at a much larger scale than conventional extraction and can use large amounts of both land area and water (average well pad = 1.2−2.7 ha, HVHF each well uses 11−30 million liters of water).[6−9] In addition to water use during HVHF, wells also produce large quantities of "produced" waters throughout the life of the well, which are often highly saline and are the single largest waste stream associated with natural gas production.[10] The potential of the extraction process to contaminate groundwater sources has received much public interest and concern, and surface and groundwater are often linked.[11−13] However, the disposal of waste materials (flowback (water that flows back to surface after HVHF and primarily consists of the fluid used to fracture the shale), produced waters, and drill cuttings)[7,13−15] and other potential effects to the landscape and ecosystems are also are of concern and have received much less attention from scientists or the public.[9,16−18]

Potential effects on terrestrial and aquatic ecosystems can result from many activities associated with the extraction process and the rate of development, such as road and pipeline construction, well pad development, well drilling and fracturing, water removal from surface and ground waters, establishment of compressor stations, and by unintended accidents such as spills or well casing failures[19,20] (Table 1). Many of these activities, such as road building and land clearing, are not unique to this process, and resulting effects on ecosystems such as sedimentation, habitat fragmentation, and shifts in community composition can be predicted from research conducted on other types of

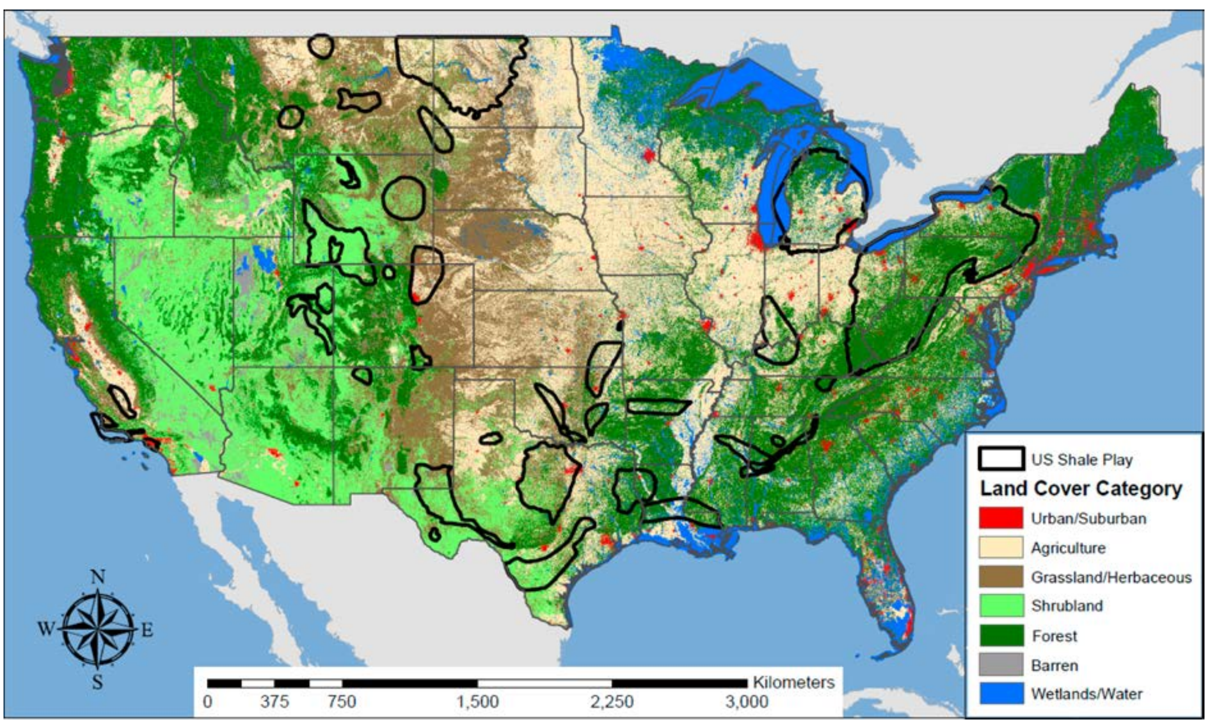

Figure 1. Landcover and habitat type overlaying shale plays in United States (National Atlas of the United States. 2005. Conterminous United States Land Cover 1992, 200-Meter Resolution). Shale plays were downloaded from the US Energy Information Administration (EIA, http://www.eia.gov/pub/oil_gas/natural_gas/analysis_publications/maps/maps.htm). Shale basins are highlighted as transparent black polygons.

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Table 1. Potential Effects of Unconventional Shale Gas Development on Terrestrial and Aquatic Ecosystems Showing the Suspected Relationship between the Oil and Gas Development, Its Effect on the Habitat and the Resulting Effect on Individual Species, Populations and Communities. See Text for Details

unconventional shale exploration and development activity	key characteristics of development activity	potential effects on habitat	potential effects on associated species and communities
initial exploration, seismic testing	heliportable units, tracked vehicles, seismic detonations	erosion, potential effects to fossorial habitats but no data on this, short-term minor disturbance to landscape, habitat fragmentation (site specific)	mostly short-term effects but will vary with species, habitat, density and width of seismic lines, and habitat recovery time. for some species, such as forest amphibians, these lines may act as barriers to movement. disturbance to fossorial species may also occur.
well pad construction	leveling of terrain, extensive earth moving activities, construction of new roads or expansion of existing roads, heavy truck traffic, import of stabilizing materials	habitat conversion and fragmentation, increased disturbance, soil compaction, erosion, runoff and sedimentation, effects on water quality	direct mortality, shifts in community composition with a trend of generalist[a] species increasing and habitat specialists[b] declining, spread of invasive species, loss of rare species or habitats
well drilling	large amounts of equipment including drilling mast, drill pipe, drill bits, drilling mud, production and storage of drill cuttings, lights to enable activity 24 h per day	local effects of noise and light pollution, high levels of traffic and disturbance, norms (naturally occurring radioactive material) when present in drill cuttings, minor use of water	local disturbance to wildlife from noise and light pollution, disturbance and potential for increased mortality associated with heavy truck traffic, potential contamination from improper disposal of drill cuttings
hydraulic fracturing	injecting water mixed with sand and chemicals into a well under high pressure to open fractures in shale and release gas; use of 11−30 million liters of water per well, impoundments or tanks to store water, heavy truck traffic to transport water, sand extraction, trucks to transport sand, trucks with chemicals for fracking, frac control van, lights to enable activity 24 h per day, temporary storage for flowback water	reduction in water quantity and ecological flows dependent on site and time of year, altered flow regimes if temporary dams are used, noise and light pollution, air pollution from diesel engines, local disturbance from sand extraction off site, potential on or offsite contamination from accidental spills or improper storage or disposal of flowback water	shifts in community composition with a trend of generalist species increasing and habitat specialists declining, spread of invasive species through unintentional introductions, direct mortality from interactions with increased traffic, direct mortality to species sensitive to changes in water quality or quantity
production	production, transport and storage of gas, wastewater production, loss of stray gas, infrastructure to move gas including compressor stations and pipelines	habitat conversion and fragmentation from pipelines; soil compaction, erosion and sedimentation; negative effects on water quality from runoff into streams; local and off site contamination from accidental spills or improper disposal of wastewater; potential for earthquakes from deep well injection ; noise pollution from compressor units	shifts in community composition with a trend of generalist species increasing and habitat specialists declining, spread of invasive species, direct mortality to species sensitive to changes in water quality or quantity, acoustic masking for species that communicate by sound, avoidance of noisy areas near compressor stations

[a]Generalist, A species that is able to use a wide range of habitats and is tolerant of a wide range of environmental conditions. [b]Specialist, A species with very specific habitat requirements and tolerant of a norrow range of habitat conditions.

anthropogenic activities. Other potential effects, for example, surface and groundwater contamination from accidental release of fracturing and drilling chemicals or waste materials, are likely novel to the oil and gas shale extraction process. Furthermore, some plays have an associated history of oil and gas development before the current expansion of unconventional oil and gas activities. The cumulative effect of these potential stressors will depend in large part on the rate of development including new infrastructure and the local extent of development in a region. Depending on extent of development, oil and gas extraction has the potential to have a large effect on associated wildlife, habitat and aquatic life.

We describe potential effects of shale development on wildlife and aquatic life across the contiguous US by reviewing existing studies that have directly examined this issue. Because these studies are rare, we also review studies associated with other activities that produce similar types of disturbance and that have long been studied. Due to space limitations and literature available on the subject, this review is weighted more heavily toward effects on ecosystems from surface water issues than from factors affecting groundwater. We further identify factors associated with at risk species or communities and conclude by highlighting monitoring and research needs to help better understand the risk to ecosystems from unconventional shale oil and gas development.

■ EFFECTS OF SHALE OIL AND GAS DEVELOPMENT ON ECOSYSTEMS

Healthy ecosystems provide numerous services including provisioning services such as food, fiber, habitat, and water; regulating services such as water purification and climate regulation; cultural services including recreational opportunities, ecotourism and aesthetics; and supporting services such as soil formation.[21] Increasing demands on ecosystems can degrade them and reduce their ability to provide these services, for example, Barnes et al. 2009.[22] The challenge we currently face is to extract oil and gas resources in a manner that ensures ecosystem sustainability. This will require an understanding of how development will impact ecosystems and a willingness to invest in the research, monitoring and management to reduce negative impacts.

Exploration and development of unconventional shale resources affects ecosystems in multiple ways (Table 1). Shale oil and gas development changes the landscape. Land is cleared for pad development and associated infrastructure, including pipelines, new and expanded roads, impoundments, and compressor stations,[6,23−25] and much of this exploration and development is occurring in relatively undeveloped landscapes. Seismic testing, roads, and pipelines bisect habitats and create linear corridors that fragment the landscape. Shale oil and gas development also can change aquatic resources. Recent reviews have identified three main potential stressors to surface waters: changes in water quantity (hydrology), sedimentation, and water quality.[8,17,26] The effects of shale development will likely differ among regions, habitats, and species, but there are some broad trends that occur; which we highlight here. Some effects, such as habitat fragmentation, are associated with multiple stages of the process and can affect both terrestrial and aquatic habitats whereas others, such as water withdrawals and subsequent changes in water quantity, are primarily associated with one stage of the extraction process and primarily affect aquatic environments.

Habitat Loss and Fragmentation. Habitat loss and fragmentation are closely intertwined, with loss of habitat frequently associated with fragmentation of the remaining habitat, and fragmentation often associated with additional losses of interior or core habitats. Loss of habitat results directly from the pad footprint and associated infrastructure (Table 1). Unconventional well pads typically average 1.2−2.7 ha,[6,9] are covered in crushed stone or wooden mats to support heavy equipment, and often have an impervious liner to contain spills. They can therefore be considered nonhabitat. Additional habitat loss and conversion occurs with pipelines, new and expanded roads, impoundments, and other infrastructure resulting in habitat conversion of an estimated 2.9−3.6 ha per pad.[6,24] Although habitat loss is usually not a direct cause of mortality, construction and development that occurs during the breeding season may be associated with direct mortality and loss of recruitment for birds nesting on site.[27]

Habitat fragmentation is one of the most pervasive threats to native ecosystems and occurs when large contiguous blocks of habitat are broken up into smaller patches by other land uses or bisected by roads, transmission lines, pipelines or other types of corridors. Habitat fragmentation is a direct result of shale development with roads and pipelines having a larger impact than the pads[6,24] (Table 1). For example, in Bradford and Washington counties Pennsylvania, forests became more fragmented primarily as a result of the new roads and pipelines associated with shale development, and development resulted in more and smaller forest patches with loss of core forest (forest >100 m from an edge) at twice the rate of overall forest loss.[24] Pipelines and roads not only resulted in loss of habitat but also created new edges. Similar results have been shown in other studies.[6,25]

A number of ecological changes are associated with fragmentation, including changes in patch size and isolation, light, moisture, and temperature, which directly and indirectly affect populations and communities.[28,29] Fragmentation has been associated with spread of invasive plant species (Table 1), many of which are associated with disturbance and can have negative impacts on native habitat quality.[30,31] Habitat fragmentation can have negative effects on wildlife populations and communities (Table 1) through direct habitat loss or indirectly through changes that occur on adjacent habitats and land uses associated with them. Negative effects of fragmentation are best documented for forest habitat,[32−34] but have also been reported for other native habitats currently experiencing an expansion of energy exploration, including grasslands[35] and shrub-steppe habitat.[4]

Fragmentation from linear corridors such as pipelines, seismic lines, and roads can alter movement patterns, species interactions and ultimately abundance depending on whether the corridor is perceived as a barrier or territory boundary or used as an avenue for travel and invasion into habitats previously inaccessible.[36−38] Many species of mammals and raptors as well as the parasitic brown-headed cowbird (*Molothrus ater*) use linear corridors for movement and hunting, potentially resulting in increased levels of predation and parasitism. In Alberta, wolves (*Canis lupus*) use of seismic lines and corridors has resulted in an increased predation risk to woodland caribou (*Rangifer tarandus caribou*) a species of conservation concern.[38,39] In forest habitat, cowbird abundance is positively associated with proximity to linear edges and corridors.[40,41]

Aquatic ecosystems can be similarly affected by habitat fragmentation. Many riverine taxa use streams as corridors for dispersal. Roads and pipelines that cross streams, especially when used in combination with culverts, often create barriers to dispersal.[42,43] If not designed correctly, these barriers can

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separate upstream and downstream populations, creating isolated populations.[44] Fish are particularly sensitive to these barriers because they are restricted to stream corridors for dispersal.

Human Activity and Disturbance. Shale oil and gas exploration and development have pulses of activity particularly associated with pad and road construction, well drilling and HVHF, and waste removal (Table 1). Truck traffic can be heavy, particularly in areas where water is being trucked in for hydraulic fracturing and produced water is being trucked out. For example, the New York State Department of Environmental Conservation estimates that development of one horizontal well requires over 3300 one-way truck trips.[45] This is a concern because roads of all types have a negative effect on wildlife through direct mortality, changes in animal behavior, and increased human access to areas, and these negative effects are usually correlated with the level of vehicular activity.[46−48] Even after a well is drilled and completed, new roads and pipelines provide access for more people, which results in increased disturbance.

Habitat alteration and human activity associated with unconventional oil and gas development are known to affect mule deer habitat selection and use[49] as well as migratory behavior.[50,51] In Wyoming, Sawyer et al.[51] found that mule deer migratory behavior was influenced by disturbance associated with coal bed gas development and observed an increase in movement rates, increased detouring from established routes, and overall decreased use of habitat along migration routes with increasing density of well pads and roads. Altered habitat use and behavior by mule deer in response to unconventional oil and gas development is of particular interest in the western U.S. where mule deer populations are stable or declining in most states[52] and there is extensive overlap between potential oil and gas development areas and mule deer distributions.[49,53]

Noise Pollution. Exploration and development of the shale resource is associated with both short-term and long-term increases in noise. In the short term, site clearing and well drilling, HVHF, and construction of roads, pipelines and other infrastructure are a limited time disturbance similar to disturbance and sound associated with clearing land and home construction (Table 1). Depending on number of wells drilled, construction and drilling can take anywhere from a few months to multiple years. Compressor stations, which are located along pipelines and are used to compress gas to facilitate movement through the pipelines, are a long-term source of noise and continuous disturbance (Table 1). Because chronic noise has been shown to have numerous costs to wildlife,[54] compressors have potential to have long-term effects on habitat quality.

For many species of wildlife, sound is important for communication, and noise from compressors can affect this process through acoustical masking and reduced transmission distances. Studies on effects of noise from compressors on songbirds have found a range of effects including individual avoidance and reduced abundance, reduced pairing success, changes in reproductive behavior and success, altered predator−prey interactions, and altered avian communities, for example, refs 55−59 Greater sage-grouse (*Centrocercus urophasianus*) gather at leks where males display in order to attract females. Lek attendance declined in areas with chronic natural gas-associated noise and, experimentally, sage-grouse were shown to experience higher levels of stress when exposed to noise.[60,61]

Changes in Water Quantity. Hydrology is an important variable that controls many instream processes such as population persistence and community structure,[62] thus changes to its natural regime has a high potential to affect resident species. Overall altered hydrologic regimes may result in less available habitat, especially critical spawning habitat for resident species.[63] Additional possible effects of altered hydrology to biota include desiccation of immobile taxa (fixed cased macroinvertebrates) and life stages (egg masses) and stranding of taxa in isolated pools. Shale development activities have potential to alter the quantity of water in streams and rivers through direct and indirect means, thereby altering the natural flow regime of streams and rivers.

When freshwater is used as the base medium for the hydraulic fracturing of shale resources, a direct stress to local streams and rivers is possible (Table 1). Hydraulic fracturing each well requires 11−30 million liters of water on average,[8] water that may come from nearby surface and groundwater sources.[64] Although the relative amount of water use is small compared to other consumptive uses (e.g., agriculture and thermo-electric power), water extraction pace and location is different and likely more important for resident taxa than the total sum withdrawn. Multiple wells may be developed on a single well pad further increasing the potential hydrologic stress on local systems. If taken during low flow or drought conditions or from small, headwater or ephemeral systems, these withdrawals increase loss of important habitat.[65,66] Development activities sometimes also include deployment of temporary dams (e.g., cofferdams) that alter local flow regimes changing them from a lotic to lentic environment. Creation of impermeable water storage reservoirs can also alter watershed hydrology by reducing infiltration.

Alternatively, in arid environments, large volumes of produced water discharged into small tributary streams can result in a change from ephemeral to perennial streams.[67] Native species in these environments have evolved to survive highly variable stream hydrologic and thermal regimes. Additionally, use of ephemeral stream channels to carry produced waters to larger receiving streams can result in destabilization and headcutting of stream channels.

Extraction activities also may alter instream flow indirectly, which could affect surface runoff and generate flashier hydrographs. For example, construction sites have higher soil bulk densities than forested areas,[68] which likely decrease infiltration rates. Salts in produced waters can also alter soil structure and reduce soil infiltration by replacing calcium with sodium.[69] Unpaved roads elevate runoff rates[70] and can alter timing and volume of streamflow, water chemistry and sediment loads, and channel morphology.[71]

Changes in Water Quality. Changes in water quality can affect stream ecosystems. Sediment runoff has been documented from well pads[72−74] (Table 1), and stream turbidity has been positively correlated to well density.[17] Extraction of shale resources also includes development of access roads, many of which are unpaved and which previous research on forestry activities has shown to increase risk of sedimentation in receiving water bodies.[70,75] The effects of sediment and siltation on streams are well-known and include loss of habitat and sensitive species; abrasion of periphyton; covering of periphyton, plants and egg masses; reduced feeding efficiency of benthic macroinvertebrates and fish, and reduced primary productivity and fish reproductive success.[76−81] If similar sedimentation occurs as a result of shale development, similar effects should be expected.

Alterations of the earth's crust can mobilize large concentrations of trace metals present in geologic formations. Research related to historical activities (e.g., mining) demonstrate instances where metals including Cu, Cd, Pb, Se, and Zn move through sediments and the foodchain to affect aquatic and terrestrial wildlife.[82−84] Furthermore, Ramirez (2005)[85] reported elevated sediment concentrations of trace metals including Cu, Cd, Ni, and Zn in holding ponds used for oil and gas activities. Those concentrations were above thresholds defined for aquatic invertebrates.

Water quality changes and contamination of water bodies are most likely to be affected by accidental release or spills, discharge of untreated or treated waters, or road application for dust and ice control. Produced waters can contain high levels of total dissolved solids (TDS), salts, metals, and naturally occurring radioactive materials.[10] Flowback waters may also contain similar constituents along with fracturing fluid additives such as surfactants and hydrocarbons.[86,87] Saline produced waters and brines are not limited to hydraulic fracturing activities, and are frequently produced during the lifetime of the well.[8] Aquatic taxa often are sensitive to increased levels of salinity;[88−93] therefore accidental releases of produced and flowback waters may have harmful effects on resident freshwater taxa.

In the Williston Basin, which includes the Bakken Shale Play and underlies portions of Montana and North Dakota, historic oil and gas drilling activities have resulted in contamination of surface and shallow ground waters from highly saline brines.[94,95] Much of this contamination is the result of outdated practices (use of unlined evaporation ponds, trenching of ponds etc.), but the contamination is persistent and movement of contaminated groundwater plumes through wetlands has been documented.[96] In some sites of historic salt contamination outside of the Williston Basin, this has resulted in replacement of native plant species with more salt-tolerant species.

The Williston basin is currently in the midst of an oil and natural gas boom. While best management practices (BMPs) used today are much more restrictive in regulation of produced waters, the rapid pace and large scale of development increases the likelihood of accidental spills and illegal dumping of brines and produced waters. In the Powder River Structural Basin (PRB) of Wyoming and Montana, the advent of coal bed natural gas technologies created an extraction boom from the late 1990s through the late 2000s. To economically extract natural gas from coal beds, water must be pumped from the well to reduce the pressure within the coal seam and allow the gas to escape.[97] This results in large quantities of produced waters. The quality of these produced waters is highly variable but within the PRB, produced waters are mildly saline and dominated by sodium bicarbonate (whereas most produced waters are predominantly sodium chloride).[98] As is often the case with emerging technologies and rapid resource exploitation, regulatory requirements lagged behind the pace of development, and early disposal methods of produced waters included direct discharge to surface waters.[99] This led to alteration of ephemeral draws (which normally only contained water during spring runoff or storm events) into perennial streams that contained up to 100% produced water. The concentration of sodium bicarbonate exceeded 3000 mg/L in several tributary streams of the Powder River, which is nearly double the median lethal concentration (LC50) for fathead minnows, a tolerant species native to the region.[100] As the play developed, direct discharge to surface waters became a less frequently used option, with ion-exchange treatment and other disposal methods used for the vast majority of well wastewater. Restrictions placed on discharge of produced waters were not driven by water quality concerns, but by the deleterious effects of sodium enriched water on soil structure and agricultural productivity.

Similar to other plays, initial development in the Appalachian Basin outpaced management practices. For example, initial disposal of wastewaters in Pennsylvania included treatment at municipal wastewater treatment plants, which were not adequate for treating such high TDS industrial wastes. This practice is now discouraged in this state.[14] Discharge of inadequately treated wastes could contaminate surface waters.[101] For example, Ferrar et al.[102] showed high levels of salinity, Cl, Ba, Br, Sr in discharges from publically owned treatment plants that handled shale oil and gas wastewaters; concentrations of all these constituents drastically decreased following cessation of treating these wastewaters. Moreover, Olmstead et al.[103] found elevated Cl concentrations in streams below treatment facilities and a positive correlation between total suspended solids and number of wells in a watershed. Warner et al.[104] reported elevated Cl and Br in streamwater and increased Ra-226 in sediments below a brine treatment facility in western Pennsylvania. However, Skalak et al.[105] found no increases in total Ra-226 and extractable Ba, Ca, Na, or Sr in fluvial sediments below publically owned treatment works or centralized waste treatment facilities that treated shale oil and gas wastewaters. But they did report elevated Ra-226 and extractable Sr, Ca, and Na in soils in proximity to roads that experienced spreading of brines for deicing, suggesting a potential nonpoint source of these pollutants.

Elevated concentrations of selenium have also been reported in flowback waters from the Marcellus shale, and in well water associated with the Barnett shale in Texas.[106,107] While no direct toxic effects from selenium have been documented within these plays, Se can be highly toxic particularly to aquatic life.[108] The potential for human health and environmental concerns associated with Se require vigilance and additional study to ensure that metals associated with flowback waters do not enter ground and surface waters.[106,107]

Large scale accidental discharge of hydraulic fracturing fluids, produced and flowback waters, and mine cuttings associated with hydraulic fracturing activities appear to be relatively uncommon. They have occurred, however, and have resulted in gill lesions in resident fish exposed to hydraulic fracturing fluids following an accidental release in Acorn Creek, KY.[109] Additionally, the authors were unable to locate endangered blackside dace, indicating the spill may have extirpated the local population. The impacts of spills and accidents are difficult to track because of nondisclosure agreements between land owners, lease holders, and energy companies. More common exposures are likely through localized brine spills and permitted discharge of produced waters with elevated TDS.[110] For example, over 850 spills ranging from 3.7 to 190 785 L (1−50 400 gallons) were reported to the North Dakota Department of health in one year.[111] Trace metals from spilled hydraulic-fracturing fluid may have effects similar to those reported for other metals, including altered periphyton[112] and benthic macroinvertebrate assemblages[113] and potential bioavailability to fish populations.[114] Researchers have documented concentrations of As, Cd, Ni, and Zn above threshold effect concentrations for sediment-dwelling organisms in closed containment and evaporation ponds associated with hydraulically fractured wells.[85,115]

■ SPECIES AND HABITATS MOST AT RISK

The long-term effects of unconventional shale oil and gas development will depend on both the characteristics of the habitat or species and the scale of development relative to the species or ecosystem of interest. Below, we highlight several key species and habitats we believe to be particularly susceptible to shale development. Although the list is nowhere near comprehensive it serves to reflect the general types of species and habitats affected by shale development.

Core Forest Habitat and Forest Specialists. Historically much of the eastern United States and Canada were forest habitat. Today, remaining areas with large blocks of core forest (generally defined as forest habitat at least 100 m from an anthropogenic edge) provide important habitat for a variety of wildlife species, including area-sensitive or forest-interior songbirds and amphibians. Because of the large overlap between the Appalachian shale play and core forest habitat in the East, many forest species are vulnerable to development. Area- sensitive forest songbirds are primarily insect-eating Neotropical migrants, are an important component of forest ecosystems, and, as a group, many have declined in numbers in response to forest fragmentation.[32,33,116] These birds are area-sensitive because breeding success and abundance are highest in large blocks of contiguous forest, and numerous research studies have documented negative effects of fragmentation on abundance and productivity (see references above under habitat loss and fragmentation). The impact that shale development has on this group of species will depend on the scale and extent of development. By some estimates, less than 10% of potential shale gas development has occurred in the Appalachian basin.[2] If this is the case, there is the potential for a 10-fold increase in the amount of shale gas development which would likely have negative impacts on area-sensitive forest songbirds and other forest specialists.

There are no published research data specifically on effects of unconventional gas development on forest-dwelling amphibians, but it has been hypothesized that they will be negatively affected by more roads and traffic, habitat fragmentation, changes in microclimate, and localized areas of increased salinity.[18] Numerous studies have documented negative effects of roads and development on amphibian abundance, diversity, and microhabitat.[117−119] Amphibians are generally negatively impacted by roads and development because of poor dispersal abilities, increased mortality from vehicles, and microclimatic drying. In addition, many species breed in vernal ponds which are negatively affected by changes in water quantity and quality and direct disturbance. Many amphibians are also highly sensitive to road salts.[120−122] Risk is particularly high for species that have large portions of their native range underlain by shale basins (e.g., Wehrle's salamander (*Plethodon wehrlei*)).[18]

Sagebrush-Steppe Habitats and Sagebrush Specialists. Sagebrush-steppe habitats of the Western U.S. have undergone extensive declines since European Settlement due to a variety of factors including conversion to agriculture, overgrazing, introduction of invasive species and energy extraction.[4,123,124] Due to the large overlap between sagebrush habitats and oil and gas extraction and because of prior degradation of this habitat, current and future development place this habitat and associated specialists at risk. One of the most well studied species is the greater sage-grouse which has undergone major populations declines in recent decades,[125,126] and large portions of their current and historical range overlaps with shale plays and/or coal-bed methane (Figure 2). Because sage-grouse are habitat specialists and have a relatively large home range, they have been considered an umbrella species for other sage-brush inhabitants[127] and have been the focus of many studies looking at effects of energy development. The effects of gas and oil structures, infrastructure, and associated disturbance on sage-grouse include avoidance of those areas for leks and nesting, reduced survival, and reduced nest success. These effects can result in population declines, and in some cases, extirpation of populations particularly in areas with high levels of oil and gas development.[123,124,128,129] Much less is known about other sagebrush obligates, but a study in Wyoming found reduced abundance of sagebrush-obligate songbirds near roads associated with gas development whereas nonobligates showed a range of responses including higher abundance near roads.[130] In another study, abundance of sagebrush-obligate songbirds declined with increasing well density[131]

Streams and Associated Biota. The U.S. EPA found that 41.9% of wadeable streams they surveyed were classified in poor condition.[132] More recently they report 55% of all rivers and streams of the U.S. were in poor condition.[133] Large-scale development of shale resources might increase these percentages, especially if the development process is not fully understood or if effective BMPs are not available or followed. Many resident species in these systems can be affected by UOG development. One example of a species at risk is the brook trout (*Salvelinus fontinalis*), whose native southern range overlays much of the Appalachian Basin. In 2008, Hudy et al.[134] reported on the distribution and status of brook trout across a large portion of their native range classifying subwatersheds as having intact, reduced, and extirpated brook trout populations (Figure 3). They reported that subwatersheds classified with intact populations (of historical habitat, over 50% currently supports self-sustaining populations[117]) most often had forest cover over 68% and had road densities less than 2.0 km per km^2. Development of shale resources, which clears land for well pads and roads, is occurring across a large portion of the native range of brook trout, especially in Pennsylvania (Figure 3). If remaining high-quality stream reaches become unsuitable to brook trout, there may be further fragmentation of the larger meta-population. Weltman-Fahs and Taylor[26] provide a critical review of potential effects of shale development on brook trout conservation and restoration, and Smith et al.[135] proposed a decision analysis framework to anticipate cumulative effects of shale development on brook trout.

Rare, Threatened, and Endangered Species. Rare species with limited ranges are always a concern when development occurs. The fact that these species are rare means that any type of disturbance can be very detrimental to them. Globally, databases such as the IUCN Red list of Threatened Species (http://www.iucnredlist.org/) may help to further identify species at risk from oil and gas development on a global scale. Within the U.S., there are federally listed species (http://www.fws.gov/endangered/), and each state also maintains a list of species of greatest conservation need to help prioritize at risk species more locally. One of the best documented cases is the greater sage-grouse (see above). Freshwater mussels are an additional taxonomic group of interest because of already high numbers of listed species and relative sensitivity to toxicants.[136] The endangered Indiana Bat, (*Myotis sodalis*), is another example of a species where a large portions of its native range is within areas of shale development (Figure 3). Gillen and Kiviat 2012[18] reviewed 15 species that were rare and whose ranges overlapped with the Marcellus or Utica shale by at least 35%. The list included the West Virginia spring salamander (*Gyrinophilus subterraneus*), a species that is on the IUCN Red List as endangered and whose range overlaps 100% with the shale layers. It requires high quality water and is sensitive to fragmentation suggesting that this species is at great risk to oil and gas development. The list also included eight Plethodontid salamanders, a group that tends to be vulnerable because of the

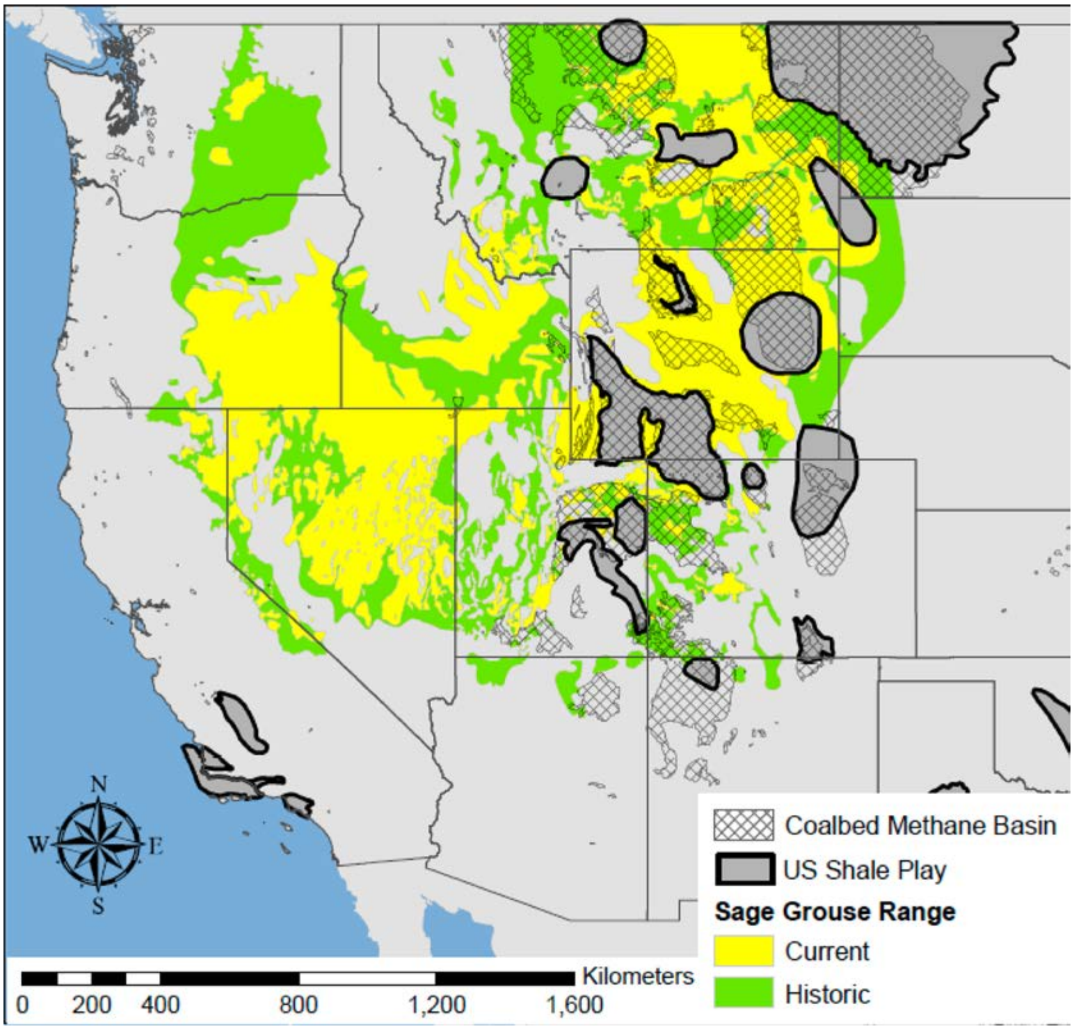

Figure 2. Current and historic distribution of sage grouse in relation to shale plays and coalbed methane deposits. Sage grouse distribution was downloaded from SAGEMAP - A GIS database for sage-grouse and shrubsteppe management in the Intermountain West, http://sagemap.wr.usgs.gov/. Shale plays and coalbed methane deposits were downloaded from the U.S. Energy Information Administration (EIA, http://www.eia.gov/pub/oil_gas/natural_gas/analysis_publications/maps/maps.htm).

overlap between their range and shale layers, their dependence on moist environments and sensitivity to disturbance.

■ RESEARCH AND MONITORING NEEDS

Together, increased development, a historical lack of peer-reviewed data, and high number of sensitive species and habitats located in shale-rich areas has sparked an increase in research on shale development in the U.S. This paper provides examples of scientific studies that can be used to aid sustainable management of this resource not only for future development of U.S. shale plays but also as information for development of shale resources across the globe. However, literature remains relatively sparse and more focused research is needed to understand ecological effects of oil and gas development.

We cannot understate the importance of baseline data. Baseline data is critically needed across all the shale plays in both terrestrial and aquatic habitats in order to detect changes as they begin to occur. Baseline data is particularly useful when collected in a manner that facilitates adaptive management and assessment of remediation and restoration plans (e.g., Smith et al. 2012).[135] Determining restoration goals and defining the baseline is often a very difficult process after the fact.

One of the first needs in understanding overall effects of shale resource development on ecosystems is to compare similarity of effects with those from other practices such as construction, urbanization and conventional oil and gas development where we have a rich research base. These studies could be expanded to focus on timing of disturbance in relation to life-stage of specific native species. Water quality stressors may have less transferability among anthropogenic stressors and therefore would require more detailed research on specific cause-effect relationships (e.g., salinity, metals, NORMs). Evaluation of current BMPs and management activities need to be scientifically validated, both in terms of the scientific basis used to develop them[137] and most importantly, their overall effectiveness. For example, in the Susquehanna River portion of the Appalachian Basin, water withdrawals for shale development are regulated by the Susquehanna River Basin Commission. Research could determine both the influence of this regulation on ecological flow needs of endemic biota and if such management practices would be applicable elsewhere.

Research and monitoring efforts need to encompass studies of potential direct and indirect effects including an effort to model effects on a large scale that can be used to understand and

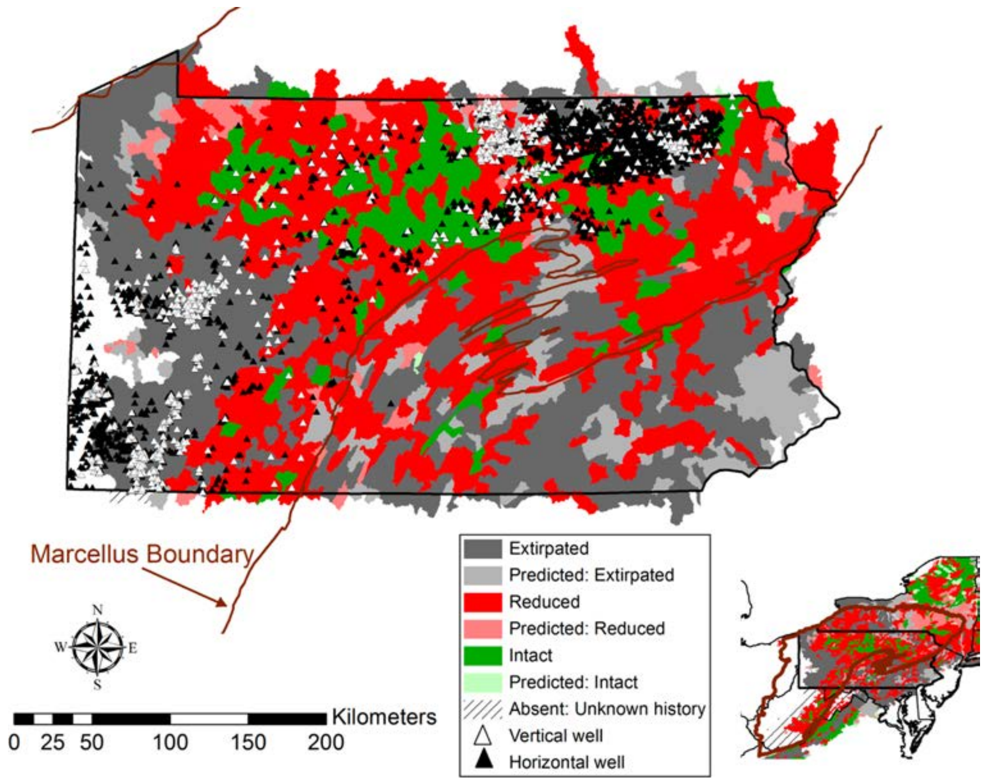

Figure 3. Map showing the spatial position of unconventional vertical ($n = 981$) and horizontal wells ($n = 6355$) for Pennsylvania, since 1 January 2000 and date of Pennsylvania spud database download (27 November 2013, http://www.depreportingservices.state.pa.us/ReportServer/Pages/ReportViewer.aspx?/Oil_Gas/Spud_External_Data) and the distribution of brook trout classification developed by the Eastern Brook Trout Joint Venture (Hudy et al. 2008; EBTJV 2012: http://easternbrooktrout.org/assessment-data/brook-trout-assessment-data/ebtjv-sub-watersheds). Inset shows state of Pennsylvania and aerial extent of the Marcellus Shale Play (EIA) in relation to the mid-Atlantic region of the U.S.

interpret changes detected in models and to define cause and effect relationships. Below, we give a broad overview of the types of research questions that could assist with informed management and policy decisions to reduce ecological risks and maintain resource values. There are five general areas of research and monitoring that could provide direct input into an adaptive management approach by reducing uncertainty in our understanding of the effects of shale gas development and risks to species and habitats: (1) spatial analyses, (2) species-based modeling, (3) vulnerability assessments, (4) ecoregional assessments, and (5) threshold and toxicity evaluations.

Spatial analyses include the mapping of resources and estimating development patterns. This area has received the most attention to date and can define use patterns, assess future and cumulative impacts given different development patterns, and evaluate trends in production and transport of wastewater.[6,14,23−25,138−140] However, with the onset of new technologies and changing energy demands these patterns can change rapidly. A need here is to update these analyses as changes occur and use them to identify the degree of overlap between key species and/or habitats and active and proposed development areas; those with most overlap would be of most need for additional studies including species-based modeling, vulnerability assessments, and threshold and toxicity evaluations.

Species-based modeling includes population biology, behavioral studies, habitat modeling, and demographic studies.[49,141−143] Demographic studies are often necessary because abundance is not always a good indicator of habitat quality and in some cases may be misleading. Species-based modeling can also be used to determine sources of mortality. Because effects vary regionally and among species, species-based models should focus initially on species and communities that are considered most vulnerable or sensitive to disturbance from oil and gas development.

Vulnerability assessments investigate overlap of sensitive species and habitats and development (e.g., brook trout assessment above; also see Gillen and Kiviat 2012[18]), the proximity of sensitive habitat to spills, for example, Preston et al.,[144] and potential direct toxicity of a spill or release on a terrestrial or aquatic population. These assessments should be incorporated into larger models that can be used to predict vulnerability across multiple scales. More focused targeted-gradient or paired watershed designs (e.g., BACI designs) can be employed to tease out effects at more local scales. Entrekin et al.[17] took a focused approach to show a positive relationship between well density and stream turbidity in seven streams in the Fayetteville shale area of Arkansas.

Ecoregional assessments consider multiple species or communities and evaluate multiple drivers of change (e.g., Bryce et al.).[145] They incorporate multiple spatial scales and identify cumulative impacts that result from energy extraction and from other types of disturbance. Mechanistic information from habitat selection studies, migration studies, demographic and behavioral studies examining response to disturbance collected during species-based modeling would provide important interpretation for these large ecoregional assessments. For example, Dauwalter[146] conducted an ecoregional assessment of oil and gas development for fish assemblages in the Colorado River Basin, Wyoming and found that assemblage structure was much more strongly related to other natural and anthropogenic factors than to oil and gas development (well density). However, he did identify several species that showed negative relationships at low levels of well density (≤0.15 wells per km^2). Studies, such as these place effects of shale development into context with other historical and ongoing anthropogenic stressors.

Threshold and toxicity evaluations can use surrogate species to identify national areas of concern and investigate resident species to document effects regionally. Farag and Harper[67] describe a multiapproach design for toxicity evaluations. This design includes laboratory and field exposure to define survival and physiological changes at the individual level to explain observed changes in populations. For example, Harper, et al.[147] defined thresholds of toxicity of saline produced waters to multiple species. Papoulias and Velasco[109] found that fish exposed to HVHF in the field showed indications of stress and had a higher number of individuals with lesions than those in reference waters. These cause-effect-type studies combined with synoptic or targeted studies will help highlight key species/biomarkers of concern (either sensitive species needing protection or indicator species/biomakers that can be used to identify a potential problem). These biomarkers will also help test the adequacy of existing BMPs. The same types of studies can be developed to test for wildlife toxicity and for threshold effects on populations related to physical changes in habitat quantity or quality.

CONCLUSION

Habitat fragmentation, effects on water quality and quantity, and cumulative effects on habitats and species of concern have already been identified as problems and are expected to increase in magnitude as shale resource development continues to expand. Our review suggests that species and habitats most at risk are ones where there is an extensive overlap between a species range or habitat type and one of the shale plays (leading to high vulnerability) coupled with intrinsic characteristics such as limited range, small population size, specialized habitat requirements, and high sensitivity to disturbance. Examples include core forest habitat and forest specialists, sagebrush habitat and specialists, vernal pond inhabitants, and stream biota. We suggest five general areas of research and monitoring that could aid in development of effective guidelines and policies to minimize negative impacts and protect vulnerable species and ecosystems including spatial analysis, species-based modeling, vulnerability assessments, ecoregional assessments, and threshold and toxicity evaluations. Responsible energy development could include provisions to avoid, reduce, and mitigate these impacts in order to conserve and protect our natural resource legacy.

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Notes

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ATTACHMENT C

STUDY 37

Rapid expansion of natural gas development poses a threat to surface waters

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Extraction of natural gas from hard-to-reach reservoirs has expanded around the world and poses multiple environmental threats to surface waters. Improved drilling and extraction technology used to access low permeability natural gas requires millions of liters of water and a suite of chemicals that may be toxic to aquatic biota. There is growing concern among the scientific community and the general public that rapid and extensive natural gas development in the US could lead to degradation of natural resources. Gas wells are often close to surface waters that could be impacted by elevated sediment runoff from pipelines and roads, alteration of streamflow as a result of water extraction, and contamination from introduced chemicals or the resulting wastewater. However, the data required to fully understand these potential threats are currently lacking. Scientists therefore need to study the changes in ecosystem structure and function caused by natural gas extraction and to use such data to inform sound environmental policy.

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Natural gas drilling has dramatically expanded with advances in extraction technology and the need for cleaner burning fuels that will help meet global energy demands. Natural gas is considered a "bridge fuel" to renewable energy resources because its combustion releases fewer contaminants (eg carbon dioxide [CO_2], nitrogen oxide [NO_x], sulfur oxide [SO_x]) than compared with that of coal or petroleum. Horizontal drilling and hydraulic fracturing ("hydrofracking" or "fracking") now allow the extraction of vast shale gas reserves previously considered inaccessible or unprofitable. Shale gas production in the US is expected to increase threefold and will account for nearly half of all natural gas produced by 2035 (EIA 2011). This widespread proliferation of new gas wells and the use of modern drilling and extraction methods have now been identified as a global conservation issue (Sutherland *et al.* 2010). Here, we describe the threats to surface waters associated with increased natural gas development in shale basins and highlight opportunities for research to address these threats.

In a nutshell:

- The construction of pipelines and roads coupled with the extraction of natural gas from shale basins may pose environmental threats
- Streams and rivers near newly drilled natural gas wells are vulnerable to sediment runoff, reduced streamflow, and possible contamination from introduced chemicals and the resulting wastewater
- Federal and state environmental regulations may not prevent or mitigate damaging effects to surface waters
- Scientific studies are needed to understand the possible environmental effects caused by activities associated with natural gas extraction

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Horizontal drilling and hydraulic fracturing

Gas-well drilling has historically used a single vertical well to access gas trapped in permeable rock formations (eg sandstone) where gas flows freely through pore spaces to the wellbore. Unlike these conventional sources, unconventional gas reservoirs are low permeability formations, such as coal beds, dense sands, and shale, that require fracturing and propping (addition of sand or other granular material suspended in the fracturing fluid to keep fractures open) before gas can travel freely to the wellbore. Hydrofracking uses high-pressure fracturing fluids, consisting of large volumes of water and numerous chemical additives, to create fractures, while added propping agents, such as sand, allow the gas to flow. Although hydrofracking was first used in the 1940s, the practice was not widely applied until the 1990s, when natural gas prices increased and advances in horizontal drilling made the technique more productive. Horizontal drilling increases the volume of rock a single well can access, thereby reducing the total number of wells required at the surface. The horizontal leg of a gas well is fractured in discrete lengths of 91–152 m, allowing up to 15 separate hydrofrack "events" along one horizontal well (Kargbo *et al.* 2010). Fracturing depth depends on target rock formations but varies from 150 m to more than 4000 m for the major shale formations in the US (US DOE 2009).

Extent of resources

The US currently has 72 trillion cubic meters (tcm) of potentially accessible natural gas – enough to last 110 years, based on 2009 rates of consumption (EIA 2011).

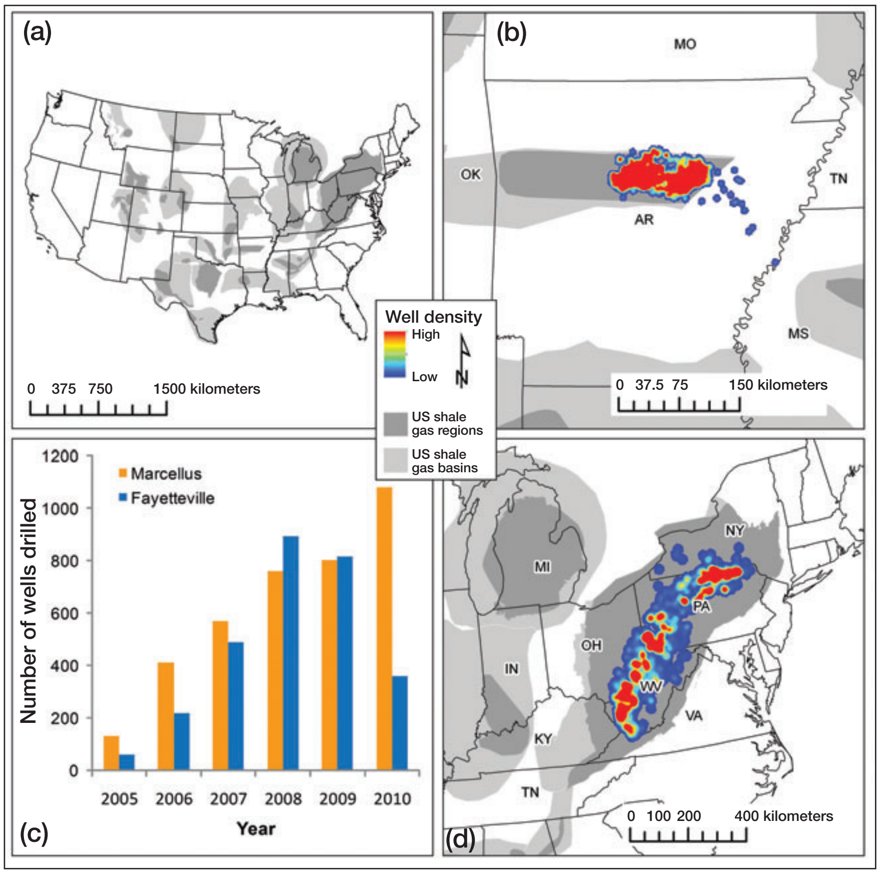

Figure 1. *(a) National map of all recognized potential areas for unconventional natural gas exploration in the contiguous US; (b) density of wells in the Fayetteville unconventional natural gas basins; (c) number of gas wells installed in the Fayetteville and Marcellus basins from 2005 to 2010; and (d) density of wells in the Marcellus shale basins. We calculated densities using the kernel density tool in ArcMap 9.3.1.*

Approximately 23 tcm of that gas is found in unconventional (ie low permeability) gas reservoirs; development of such reservoirs has increased by 65% since 1998 (US DOE 2009). There are 29 known shale basins spanning 20 states, which are expected to contribute 45% of the total US gas produced by 2035 (EIA 2011; Figure 1a). Furthermore, the US gas supply represents only a fraction of the total global estimate of potentially accessible natural gas (~459 tcm) and, outside of North America, only 11% has so far been recovered (MIT 2010). Development of potentially accessible natural gas is expected to increase with rising global demand and the transfer of drilling technologies overseas.

Threats to surface waters

The rapid expansion in natural gas development threatens surface-water quality at multiple points, creating a need to assess and understand the overall costs and benefits of extracting this resource from shale reservoirs. Gas-well development of any type creates surface disturbances as a result of land clearing, infrastructure development, and release of contaminants produced from deep groundwater (eg brines). However, the use of hydraulic fracturing poses additional environmental threats due to water withdrawals and contamination from fracking-fluid chemicals. Extraction of gas from shale formations may also produce considerably more methane (CH_4) than conventional wells and could have a larger greenhouse-gas footprint than other fossil-fuel development (Howarth *et al.* 2011). Furthermore, gas wells are often located adjacent to rivers and streams and may occur at high densities in productive shale basins, resulting in cumulative impacts within watersheds. Environmental and human health concerns associated with hydrofracking have stirred much debate, and the practice has received extensive attention from the media (Urbina 2011) and from researchers (US EPA 2004; Kargbo *et al.* 2010; Osborn *et al.* 2011; US EPA 2011; Colborn *et al.* in press). Research that addresses concerns regarding increased drilling and hydrofracking in shale basins has primarily focused on contaminants that threaten drinking water and groundwater, whereas data collection to address concerns associated with surface water and terrestrial ecosystems has largely been overlooked.

Our goal here is to provide background information on shale development in the US that may inform future ecological studies that assess the potential for environmental impacts. We use data from the Fayetteville and Marcellus shale formations to demonstrate the recent accelerated drilling activity, well proximity to streams, and well density relationships with stream turbidity. We also review other potential threats to aquatic freshwater ecosystems as a result of increased natural gas development.

Focus areas

The Fayetteville and Marcellus shale basins are among the most productive in the US. The Fayetteville shale basin underlies more than 23 000 km^2 of Arkansas and eastern Oklahoma, at a depth of 300–2000 m (Figure 1a). The number of gas wells sited in this area has increased nearly 50-fold, from 60 to 2834 wells since 2005, in a concentrated area of north-central Arkansas (Figure 1, b and c). The Marcellus shale basin spans 240 000 km^2 at a depth of 1200–2500 m and underlies six states in the upper Mid-Atlantic, including much of the Appalachian region (Figure 1d). Estimates indicate natural gas reserves in the Marcellus to be 14 tcm, or 59% of the total estimated unconventional reserves in the US (US DOE 2009). As of summer 2010, the Marcellus had 3758 natural gas wells, with projections of up to 60 000 wells being constructed in the region over the next 30 years (Johnson 2011). The Marcellus formation also underlies sensitive watersheds, such as the threatened upper Delaware River, a designated wild and scenic river that supplies drinking water to >15 million people (DRBC 2008). The rapid development of gas wells in relatively concentrated areas may increase the likelihood of ecological impacts on surrounding forests and streams.

■ Proximity of gas-well development to water resources

We initially assessed the proximity of active gas wells to water resources using state well-location data and the National Hydrography Dataset (NHD) flowlines (ie streams and rivers mapped from 1:24 000 Digital Line Graph hydrography data). Spatial analysis indicated that, for both the Fayetteville and Marcellus shale formations, gas wells were sited, on average, 300 m from streams, yet several hundred wells were located within 100 m of stream channels (Table 1). Gas wells were located, on average, 15 km from public surface-water drinking supplies, and 37 km and 123 km from public well water supplies in the Marcellus and Fayetteville shale reservoirs, respectively (Table 1). Although wells are generally constructed far from public drinking-water sources, there is potential for wastewater to travel long distances, given that many of the components of produced waters (ie a mixture of fracking fluids and natural geologic formation water flowing back out of the well), such as brines, will not settle out or be assimilated into biomass. Furthermore, the NHD underestimates the density of headwater stream channels (Heine *et al*. 2004), so our proximity measures probably underestimate the threat to streams. We therefore used geographic information system (GIS) tools to generate detailed drainage-area networks in portions of the Fayetteville and Marcellus shale reservoirs where gas wells occur at high densities. The terrain processing tools in ArcHydro Tools 9 version 1.3 (an ArcGIS extension) were used to generate drainage area lines from 10-m digital elevation models (http://seamless.usgs.gov/ned13.php) in a subset of drainage areas in each shale basin. A stream threshold of 500 (50 000 m^2) was used to define stream channels in the model. Gas-well proximity was analyzed again with a subset of modeled stream drainage areas and the same subset of NHD flowlines for comparison (Figure 2; Table 2). Active gas wells were an average of 130 m and 153 m from modeled drainage areas, as compared with 230 m and 252 m from NHD flowlines, in the Fayetteville and Marcellus shale reservoirs, respectively. Over 80% of the

Table 1. Number of unconventional gas wells drilled each year since 2005 for Arkansas, New York, Ohio, Pennsylvania, and West Virginia

	State	Total wells	Total operators	Distance to NHD flowlines (mean ± SD, range, m)	Total # (percent) of wells within: 100 m of NHD flowlines	200 m of NHD flowlines	300 m of NHD flowlines	Distance to: public water wells (mean ± SD, range, km)	public drinking-water intakes (mean ± SD, range, km)
Marcellus	PA	2091*	59	319 ± 171 (8–1172)	74 (4)	577 (28)	1141 (55)	25.83 ± 17.93 (0.32–79.60)	14.83 ± 10.06 (0.60–50.23)
Marcellus	WV	1599†	86	214 ± 143 (1–850)	409 (26)	798 (50)	1198 (75)	52.32 ± 32.81 (0.55–125.42)	11.16 ± 5.36 (0.53–33.32)
Marcellus	OH	42‡	12	230 ± 153 (46–691)	8 (19)	23 (55)	33 (79)	71.85 ± 28.29 (26.46–138.17)	14.15 ± 8.38 (1.54–29.87)
Marcellus	NY	26§	9	247 ± 182 (27–631)	9 (35)	12 (46)	14 (54)	10.47 ± 7.11 (2.58–34.19)	16.59 ± 9.06 (4.58–35.63)
Marcellus	All four states combined	3758		273 ± 168 (1–1172)	500 (13)	1410 (38)	2386 (64)	37.51 ± 28.88 (0.32–138.17)	13.27 ± 8.55 (0.53–50.23)
Fayetteville	AR	2834¶	21	353 ± 241 (7–1642)	269 (10)	900 (32)	1434 (51)	123.67 ± 11.12 (78.94–156.12)	15.15 ± 7.49 (0.66–133.43)

Notes: *PA: Pennsylvania Department of Environmental Protection Bureau of Oil and Gas (records available through 30 Sep 2010), www.dep.state.pa.us/dep/deputate/minres/oilgas/reports.htm; †WV: West Virginia Geological and Economic Survey (records available through early Sep 2010), www.wvges.wvnet.edu/www/datastat/devshales.htm; ‡OH: Ohio Department of Natural Resources Division of Mineral Resources Management (records available through 30 Sep 2010), www.dnr.state.oh.us/mineral/database/tabid/17730/Default.aspx; §NY: New York State of Environmental Conservation (records available through 30 Sep 2010), www.dec.ny.gov/energy/1603.html; ¶AR: Arkansas Oil and Gas Commission (records available through 30 Sep 2010), www.aogc.state.ar.us/ (data downloaded from: ftp://www.aogc.state.ar.us/).

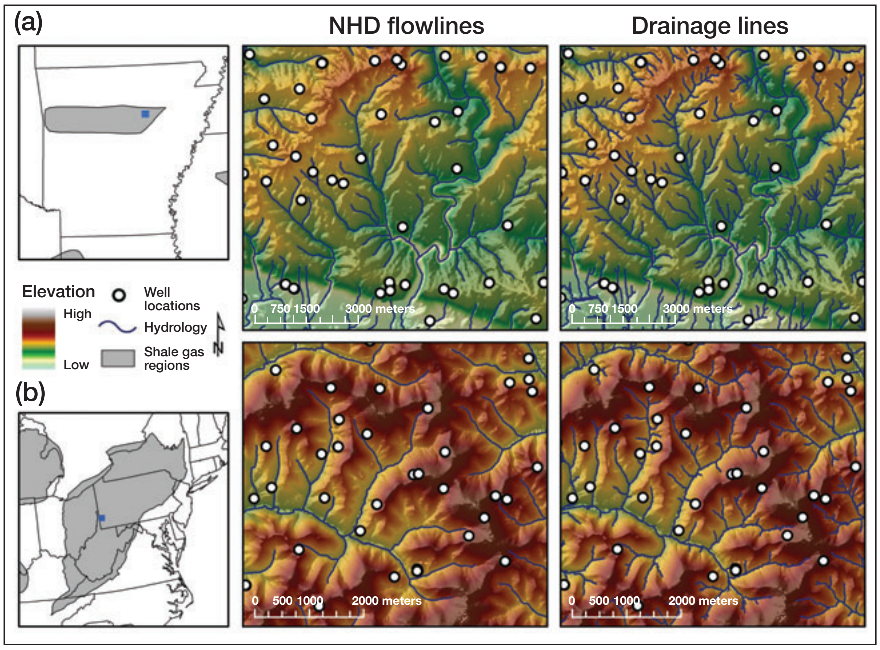

Figure 2. *Proximity of gas wells to stream channels in a subset of the Fayetteville and Marcellus unconventional natural gas reservoirs. Blue squares represent the areas modeled by GIS in the Fayetteville shale ([a] drainage area modeled and represented by the blue square was 5809 km^2) and the Marcellus shale ([b] drainage area modeled and represented by the blue square was 4041 km^2). Topographic maps are example areas that demonstrate differences between the National Hydrography Dataset and modeled drainage area networks.*

active gas wells were located within 300 m of modeled drainage areas (Table 2). Because the modeled drainage areas estimate some intermittent and ephemeral channels, the proximity of wells to stream channels (and the potential for downstream impacts) is greater than that reflected by NHD flowline data. This process may provide a more accurate assessment of potential stream impacts, particularly if shale gas development continues at its current rate. As gas-well densities continue to increase, the proximity of wells to stream channels may also increase, resulting in a greater risk of streamflow reductions from pumping, contamination from leaks and spills from produced waters or fracking fluids, and sedimentation from infrastructure development (eg pipelines and roads).

Environmental regulation

Environmental regulation of oil and gas drilling is complex and varies greatly between states. The Safe Drinking Water Act (SDWA) provides federal laws for protecting surface and groundwaters and human health, but with the exception of diesel-fuel injection, hydraulic fracturing operations are exempt as a result of the 2005 Energy Policy Act. State agencies are therefore primarily responsible for regulation and enforcement of environmental issues associated with natural gas development. The rapid growth and expansion of US gas drilling has made regulation difficult, and violations are common; in Pennsylvania alone, there were more than 1400 drilling violations between January 2008 and October 2010 (PADEP 2010). Of these, nearly half dealt with surface-water contamination and included direct discharge of pollutants, improper erosion control, or failure to properly contain wastes. In contrast, the Arkansas Department of Environmental Quality cited only 15 surface-water violations in the Fayetteville shale in 2010; however, over half of these dealt with permitting and discharge violations associated with natural gas development (ADEQ 2010). The discrepancy in the numbers of violations between states demonstrates the variable degree of regulation at the state level and is probably based on differences in regulations as well as available regulatory resources. The number and proportion of violations associated with natural gas development indicates that sediments

Table 2. Proximity of natural gas wells to stream channels modeled by terrain processing tools in ArcHydro Tools 9 (version 1.3) to generate drainage area lines from a 10-m digital elevation model (http://seamless.usgs.gov/ned13.php) as compared with well proximity to National Hydrography Dataset flowlines

		Subset		*Previous distances (in Marcellus, PA only)*		*Subset*			*Previous distances (in Marcellus, PA only)*		
		range (m)	*mean ± SD (m)*	*range (m)*	*mean ± SD (m)*	*within 100 m*	*within 200 m*	*within 300 m*	*within 100 m*	*within 200 m*	*within 300 m*
Marcellus*	Drainage area lines	4–316	153 ± 56	–	–	17%	80%	100%	–	–	–
	NHD flowlines	48–681	252 ± 114	8–1172	319 ± 171	5%	39%	70%	4%	28%	55%
Fayetteville**	Drainage area lines	0–420	130 ± 70	–	–	32%	71%	82%	–	–	–
	NHD flowlines	1–933	230 ± 136	7–1642	353 ± 241	12%	43%	61%	10%	32%	51%

Notes: *Processed for 615 of 3758 wells (16%), processed 42 of 559 HUC-12 Units containing well point locations (8%). **Processed for 2372 of 2834 wells (84%), processed 55 of 84 HUC-12 Units containing well point locations (65%).

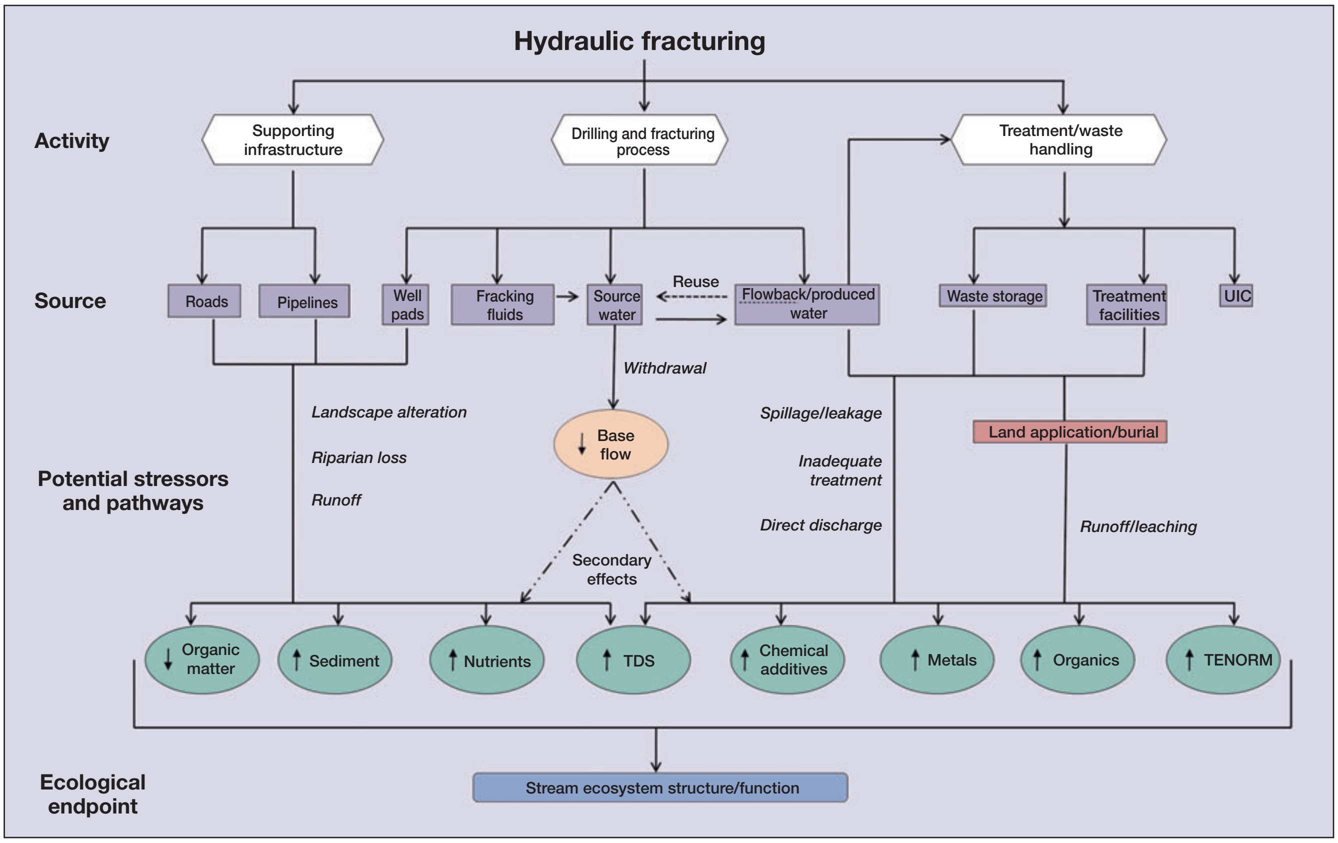

Figure 3. *Simplified diagram of potential threats due to natural gas development through coupled horizontal drilling with hydraulic fluid fracturing in unconventional natural gas reservoirs. Exposure pathways that may result in structural and functional alterations to aquatic ecosystems will vary, depending on geographic location and rigor of best management practices applied. UIC = underground injection control; TDS = total dissolved solids; TENORM = technologically enhanced naturally occurring radioactive materials. Dotted lines indicate secondary effects from gas development. "Flowback" is underlined to indicate that it may be recycled and reused.*

and contaminants associated with drilling are making their way into surface waters, and yet there are few studies examining their ecological effects. Primary threats to surface waters and potential exposure pathways (Figure 3) include sediments, water withdrawal, and release of wastewater.

Sediments

Excessive sediment levels are one of the primary threats to US surface waters (US EPA 2006) and have multiple negative effects in lotic (river, stream, or spring) food webs (Wood and Armitage 1999). Gas-well installation activities can negatively affect lotic ecosystems by increasing sediment inputs from well pads and supporting infrastructure (eg roads, pipelines, stream crossings), as well as loss of riparian area. Typically, at least 1.5–3.0 ha of land must be cleared for each well pad, depending on the number of wells per pad; where these occur in high densities, well pads can cumulatively alter the landscape. Land clearing and stream disturbance during well and infrastructure development can increase sediments in surface-water runoff (Williams *et al.* 2008), resulting in increased suspended and benthic sediments in surface waters. Nutrients, such as phosphorus, bound to these sediments may also have negative impacts on surface waters by contributing to eutrophication.

We identified seven streams in the Fayetteville shale with a variety of different well densities within their drainage areas, to test the prediction that stream turbidity would be positively related to the density of gas wells. The seven stream drainages were delineated through the use of the ArcHydro extension in ArcMap (version 9.3.1 ESRI). Using gas-well location data obtained from the Arkansas Oil and Gas Commission (ftp://www.aogc.state.ar.us/GIS_Files/), we quantified well density within each drainage area as the total number of wells divided by the drainage area. Turbidity was measured with a Hach Lamotte 2020 meter in April 2009, during high spring flow. Pearson product moment correlations identified a positive relationship between streamwater turbidity and well density (Figure 4). Turbidity was not positively correlated to other land-cover variables, but there was a strong negative correlation between turbidity and drainage area and percent pasture cover in the watershed (Table 3). These preliminary data suggest that the cumulative effects from gas well and associated infrastructure development may be detectable at the landscape scale.

Water withdrawal may alter flow regime

Surface waters may serve as sources for necessary drilling and fracking fluids – each well uses between 2–7 million

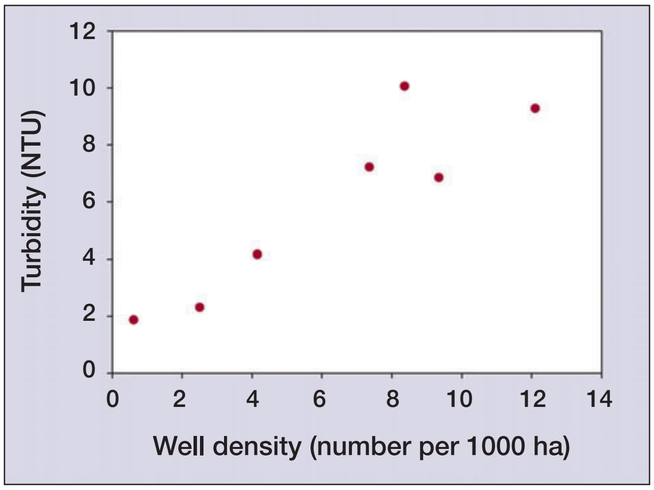

Figure 4. *Well density and stream turbidity measured in April 2009 during high flows in seven stream drainages. NTU = nephelometric turbidity unit.*

gallons (~7.5–26 million liters) of source water. Several wells may be fractured per well pad over the life span of well development, which may last several decades. This concentration of fracturing effort within a small area should compound water use. Many gas wells are installed in regions where water is already being withdrawn for agriculture, and thus may further stress the resource. Streamflow may be negatively affected if streams are dammed to create holding ponds or if water is directly extracted for the fracturing process. The rapid and concentrated extraction of water could create regional shortages during periods of drought, resulting in an altered flow regime and the further degradation of critical habitat for aquatic biota, particularly if low-order streams are primary sources. A reduction in streamflow may also result in secondary effects, such as increased contaminant concentrations and reduced downstream water quality, because less water is available for dilution.

Release of wastewaters

Surface-water contamination from hydrofracking fluids and produced water is most likely to occur during hydrofracking or treatment and disposal processes, when the potential for accidental spills and leaking is greatest. Contamination from hydrofracking wastes can also occur through inadequate waste treatment practices, improper waste storage, inadequately constructed impoundments or well casings, and improper disposal of solid wastes (eg in poorly lined impoundments that are buried onsite) that may leach into nearby surface waters. Wastewater impoundment ponds can therefore also pose a threat to wildlife and livestock.

Table 3. Pearson product moment correlations[*] and associated *P* values between turbidity (NTU) and other landscape-level variables, including land cover (Gorham and Tullis 2007) and drainage area

Correlates	*r*	*P* value
Well density	0.91	0.003
Drainage area	–0.86	0.01
Low-impact urban	0.35	0.44
Wood/herbaceous	–0.63	0.12
Forest	–0.36	0.42
Pasture	–0.88	0.008

[*]Quantifies the strength of the directional relationship. Analyses were run in SigmaPlot 11.

Fracturing fluids typically include a combination of additives that serve as friction reducers, cross-linkers, breakers, surfactants, biocides, pH adjusters, scale inhibitors, and gelling agents (NYSDEC 2010). The aim of additives is to achieve an ideal viscosity that encourages fracturing of the shale and improves gas flow, but discourages microbial growth and corrosion that can inhibit recovery efficiency (US DOE 2009). Composition of the fracturing fluids can vary greatly among wells and shale formations. Specific content is often proprietary, although some states require disclosure of constituents and companies may voluntarily register the chemicals they use with regulatory agencies. A recent Congressional investigation revealed that, over a 4-year period, 14 leading gas companies used over 2500 hydrofracking products that contained 750 different chemicals, 29 of which were highly toxic or known carcinogens. Fracturing fluids used over the period totaled 780 million gallons or ~2.9 billion liters (not including dilution water), and included lead, ethylene glycol, diesel, and formaldehyde, as well as benzene, toluene, ethylbenzene, and xylene compounds (US House of Representatives Committee on Energy and Commerce 2011). The volume of fracking fluids recovered is also highly variable, but unrecovered amounts can be substantial. Only 10–30% of fracture fluids are typically recovered from wells in portions of the Marcellus shale (NYSDEC 2010); there is currently no information on the fate and transport of the unrecovered chemicals.

Produced waters pose a threat to surface waters because they typically contain not only fracking additives but also elevated levels of metals, dissolved solids (eg brine), organics, and radionuclides that occur naturally in deep groundwaters. Onsite waste impoundments or evaporation ponds could overflow, spill, or leach into groundwater and contaminate nearby streams. Even after treatment, total dissolved solids (TDS) in produced waters are very high and remaining salts are often disposed of through land application or used as road salts, which are known to enter surface waters and contribute to increased stream salinization (Kaushal *et al.* 2005). Recovered wastewaters are most often transported offsite for deep-well injection or to a domestic wastewater treatment plant (WWTP) and/or conventional waste treatment facility. After fracturing, initially recovered flowback water is sometimes reused as fracking fluid for other wells. Reuse of recovered fluids is becoming more common, but still requires a substantial amount of fresh water because of low recovery volumes and the need to dilute flowback water containing high concentrations of chlorides, sulfates, barium, and other potentially harmful substances. Domestic WWTPs are not capable of treating the high TDS (5000 to >100 000 mg L^{-1}) typical of recovered wastewater. Many WWTPs have therefore been forced to limit their intake of recovered hydrofracking waste to remain in compliance with effluent limitations (Veil 2010). Industrial WWTPs are better equipped to treat recovered wastes using reverse osmosis, filtration, or chemical precipitation, but such facilities are costly and not widely available. Therefore, although billions of liters of produced water are being generated annually on a national scale by hydrofracking (Clark and Veil 2009), water treatment options are limited, and the potential ecological impacts of wastes on terrestrial and aquatic ecosystems are not well studied.

■ Challenges and potential for new research

Quantifying the effects of natural gas development on surface waters in shale basins is difficult because multiple companies often work in the same geographical area and use different fracturing techniques (eg varied and often proprietary composition of fracturing fluids), resulting in uncoordinated timing of infrastructure development and well fracturing. In addition, the degree to which these companies adhere to best management practices, such as buffer strips and erosion control devices, varies among companies as a result of the differing regulations among states and agencies. Furthermore, wells occur across human-impacted watersheds with characteristics that may confound our ability to attribute effects from gas-well development.

Most studies that examine the effects of sediments on biological communities focus on shifts in abundance, biomass, diversity, or community composition (Wood and Armitage 1999); few studies have analyzed how sediments alter species' roles and their interactions (but see Hazelton and Grossman 2009). In addition, contaminant effects are often assessed through single-species laboratory acute and chronic toxicity tests with standardized test organisms (eg *Daphnia*, fathead minnows [*Pimephales promelas*]; Cairns 1983) and with single contaminants. Studies are therefore needed to assess the toxicity of contaminant mixtures (eg produced water and fracking fluids) and their effects on more complex communities and ecosystems, to predict effects in the real world (Clements and Newman 2002). Sediment and contaminants associated with recovered wastewater will likely affect organism behavior and alter ecological interactions at sublethal levels (Evans-White and Lamberti 2009). Reductions in feeding efficiencies (Sandheinrich and Atchison 1989) can lead to negative effects on reproduction (Burkhead and Jelks 2001) and growth (Peckarsky 1984), and may alter the magnitude or sign (+ or –) of species' effects, causing changes in community structure. Ecologists studying the environmental effects of natural gas extraction can therefore contribute to scientific understanding by examining the effects of sediment and contaminants from natural gas development on species and community interactions.

In addition to the need for traditional bioassessments, the inevitable alteration in land use that will occur as a result of rapid and expanded drilling offers a template for conducting novel experiments in an ecosystem context. Ecosystem functions, such as decomposition rates, are affected by multiple abiotic and biotic factors, making them well-suited for detecting large-scale alterations (Bunn *et al.* 1999). For example, reduced streamflows, contaminants from produced wastewater and fracking fluids, and elevated sediment inputs would alter ecosystem functions, such as whole-stream metabolism, decomposition of organic matter, and accrual of macroinvertebrate biomass over time. However, it is not known how natural gas development could influence biological processing rates. The potential effects may stimulate or inhibit specific ecosystem functions. For example, excessive sedimentation or chemical contamination associated with natural-gas-well development could stimulate macroinvertebrate production by expanding habitat for tolerant, multivoltine (species that produce several broods per season) taxa (Stone and Wallace 1998) or lead to a decline in production by eliminating sensitive taxa representing a majority of community growth and/or biomass (Woodcock and Huryn 2007). A move to incorporate ecosystem functions into mainstream biological assessment and restoration protocols is currently underway (Fritz *et al.* 2010), yet few studies have been conducted to inform their implementation and interpretation in the context of concurrent structural changes (Young and Collier 2009). The rapid expansion of gas development across the US could provide a framework for the implementation of concurrent structural and ecosystem experiments to inform process-based ecological assessment. Furthermore, ecological studies relating to natural gas extraction could be combined with similar studies for surface mining (Fritz *et al.* 2010; Bernhardt and Palmer 2011), to gain a more holistic view of the environmental costs associated with fossil-fuel extraction.

The distinct elemental composition and isotopic signatures of produced water provide unique opportunities for tracer studies that could indicate aquatic system exposure. Stable isotopes of strontium and carbon have been used to trace water from coalbed natural gas production wells to surface waters and hyporheic zones (Brinck and Frost 2007). Osborn *et al.* (2011) used isotopes of water, carbon, boron, and radium to test for hydraulic fracturing contamination of shallow aquifers overlying the Marcellus and Utica shale formations in Pennsylvania and New York, respectively, and found significant changes in CH_4 concentrations in drinking-water wells near locations where gas wells have been drilled. Limited research has also suggested that CH_4-derived carbon is assimilated into stream food webs (Kohzu *et al.* 2004; Trimmer *et al.* 2010). Many gas-bearing geological formations also contain elevated levels of naturally occurring radioactive materials, such as radon (^{222}Rn) and radium (^{226}Ra, ^{228}Ra), that can be used as hydrological tracers (Genereux and Hemond 1990). The extent to which metals, organics, or other contaminants from the drilling and hydrofracking process may ultimately enter aquatic and terrestrial food webs remains unknown.

■ Conclusions

Natural gas exploration will continue to expand globally. In addition to the potential threats to groundwater and drinking-water sources, increasing environmental stress to surface-water ecosystems is of serious concern. Scientific data are needed that will inform ecologically sound development and decision making and ensure protection of water resources. Elevated sediment runoff into streams, reductions in streamflow, contamination of streams from accidental spills, and inadequate treatment practices for recovered wastewaters are realistic threats. Gas wells are often sited close to streams, increasing the probability of harm to surface waters, and preliminary data suggest the potential for detectable effects from sedimentation. Regulations that consider proximity of natural gas development to surface waters may therefore be needed. Further ecological research on impacts from developing natural-gas-well infrastructure are sorely needed, and will inform future regulatory strategies and improve our understanding of the factors affecting community structure and ecosystem function.

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ATTACHMENT C

STUDY 38

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Unconventional oil and gas spills: Materials, volumes, and risks to surface waters in four states of the U.S.

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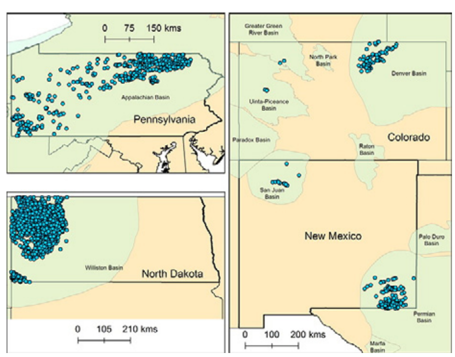
HIGHLIGHTS

- Analyzed data from 6622 spills from horizontal UOG wells in four U.S. states
- Wastewater, crude oil, HF solution and drilling waste were most often spilled
- Average distance of spills to the nearest stream was smallest in Pennsylvania
- Some spills in all states occurred within current surface water setback regulations
- Pennsylvania spills occurred in watersheds of higher importance to drinking water

GRAPHICAL ABSTRACT



Distribution of spills attributed to unconventional oil and gas wells by state. Light green polygons indicate shale basins (basin nomenclature and shapefile from USEIA (2011)).

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ABSTRACT

Extraction of oil and gas from unconventional sources, such as shale, has dramatically increased over the past ten years, raising the potential for spills or releases of chemicals, waste materials, and oil and gas. We analyzed spill data associated with unconventional wells from Colorado, New Mexico, North Dakota and Pennsylvania from 2005 to 2014, where we defined unconventional wells as horizontally drilled into an unconventional formation. We identified materials spilled by state and for each material we summarized frequency, volumes and spill rates. We evaluated the environmental risk of spills by calculating distance to the nearest stream and compared these distances to existing setback regulations. Finally, we summarized relative importance to drinking water in watersheds where spills occurred. Across all four states, we identified 21,300 unconventional wells and 6622 reported spills. The number of horizontal well bores increased sharply beginning in the late 2000s; spill rates also

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increased for all states except PA where the rate initially increased, reached a maximum in 2009 and then decreased. Wastewater, crude oil, drilling waste, and hydraulic fracturing fluid were the materials most often spilled; spilled volumes of these materials largely ranged from 100 to 10,000 L. Across all states, the average distance of spills to a stream was highest in New Mexico (1379 m), followed by Colorado (747 m), North Dakota (598 m) and then Pennsylvania (268 m), and 7.0, 13.3, and 20.4% of spills occurred within existing surface water setback regulations of 30.5, 61.0, and 91.4 m, respectively. Pennsylvania spills occurred in watersheds with a higher relative importance to drinking water than the other three states. Results from this study can inform risk assessments by providing improved input parameters on volume and rates of materials spilled, and guide regulations and the management policy of spills.

1. Introduction

Development of oil and gas from unconventional shale sources (unconventional oil and gas, or UOG) has dramatically increased over the past ten years in large part due to the combination of horizontal drilling and hydraulic fracturing. Horizontal drilling refers to the process where a wellbore aligns horizontally with the target formation, thus increasing contact with the reservoir, and hydraulic fracturing refers to the process that stimulates oil and gas within the reservoir by expanding fractures in shale through injection of fracturing fluid (i.e., water, proppant and chemicals) (USDOE, 2009). The U.S. is currently the leader in developing UOG resources, where from 2000 to 2016 daily production of shale gas (dry) increased by 20-fold (2.2 to 44.0 billion cubic feet) and tight oil increased by >10-fold (0.4 to >4.5 million barrels) (USEIA, 2016). Other countries are beginning to commercially produce oil and gas from shale and low-permeability formations (USEIA, 2015), and by 2040, unconventional gas production is projected to triple to account for almost a third of global natural gas production (IEA, 2015). Given the rapid, recent development of UOG, data are scarce on its long-term environmental impacts, and there is a need to better quantify risk to people and nature (Finkel and Hays, 2013; Small et al., 2014; Souther et al., 2014; Werner et al., 2015).

UOG development can affect species, ecosystems, and the services they provide to people. In central North America, estimates suggest that oil and gas development (including coal bed methane) reduced net primary productivity, an important measure of a region's ability to provide ecosystem services, by ~4.5 Tg of carbon from 2000 to 2012 (Allred et al., 2015). Further, land application of hydraulic fracturing fluid resulted in leaf drop and 56% mortality of trees where the application occurred (Adams, 2011). Forest interior bird counts increased with distance from a well pad in Pennsylvania (Barton et al., 2016), abundances of sagebrush songbirds decreased with increased well density in Wyoming (Gilbert and Chalfoun, 2011), and mule deer have been documented to avoid well pads with active drilling by at least 800 m in Colorado (Northrup et al., 2015). In Kentucky, an accidental release of hydraulic fracturing fluid into a stream increased gill lesions and other indicators of stress in fish (Papoulias and Velasco, 2013), and in Pennsylvania, juvenile mussels below a brine treatment plant had lower survival rates than mussels located above the plant (Patnode et al., 2015). Streambed microbial diversity was lower below an oil and gas waste injection plant in West Virginia (Akob et al., 2016), and water downstream from this site had higher endocrine-disrupting activities than reference water (Kassotis et al., 2016). Despite the emerging evidence, studies establishing causal relationships between environmental changes and UOG activities are scarce; this is particularly true for spills and releases of materials used in and produced by UOG development.

Summary reports on UOG spills are starting to emerge; however, they are typically restricted to a single state, short on detail regarding materials spilled or reasons for spills, or are characterized by a small sample size. For example, the Colorado Oil and Gas Conservation Commission (COGCC, 2014) reported that equipment failure and human error were the two main causes of spills, most spills occurred during the production stage, process piping, pipelines and tanks were the main sources of spills, and the volume of 12% of the spills were >100 barrels (15,900 L); however no detailed analysis on spilled material was presented. Brantley et al. (2014), using the Pennsylvania Notice of Violation (NOV) database, reported that one-fifth of wells were given a non-administrative violation from 2005 to 2013, and Rahm et al. (2015) reported that Pennsylvania NOVs (2007–2013) related to spills and erosion were the most common NOV issued. Neither study, however, conducted a detailed analysis on volumes or materials spilled or their potential impacts to surface waters in Pennsylvania. Finally, the U.S. Environmental Protection Agency (USEPA, 2015a) reviewed over 36,000 spill records from nine states but was able to confidently identify only 457 incidents associated with hydraulic fracturing (~12,000 contained insufficient information and ~24,000 were not related to hydraulic fracturing). The USEPA reported most spills were small (< 1000 gal, 3785 L), flowback and produced waters were the most commonly spilled material, human error was the most common cause of a spill, storage units were the common source of spills, and 300 of the spills reached an environmental receptor; however, the study did not include spills that occurred during the drilling stage.

The objectives of this study were to characterize the volumes and compositions of the materials spilled from horizontal, hydraulically fractured oil and gas wells, and evaluate the risk that spills posed to streams and surrounding watersheds important to human drinking water. Our first objective aimed to fill the knowledge gap on the materials and volumes spilled during UOG development. Our second objective focuses on streams because they provide habitat that supports a high level of biodiversity (Meyer et al., 2007), are particularly vulnerable to UOG development due to their tight coupling with upstream catchments (Hynes, 1975), and are sensitive to small changes in catchment conditions from anthropogenic activities (Maloney et al., 2012). Further, over 1/3 of the U.S. population uses public drinking water systems that rely, at least in part, on intermittent, ephemeral or headwater streams (USEPA, 2009). The spatial position of anthropogenic activities within the catchment often affects these relationships (King et al., 2005), which is especially important for UOG because wells are frequently located in close proximity to streams (Entrekin et al., 2011). We therefore evaluated the risk of spills to streams by quantifying the spatial position of spills to the nearest stream and how these distances related to current setback regulations. Because a large population relies on surface water for domestic use, our second objective also explored risks to drinking water using the U.S. Forest Service's Forest to Faucets data set. We provide a broad analysis of spill features to improve understanding of the potential environmental risks of spilled materials from UOG development and to inform management practices and policy formulation.

2. Study site and methods

2.1. Study sites and setback regulations

We sampled state databases on spill records for four states (Colorado – CO, New Mexico – NM, North Dakota – ND, and Pennsylvania – PA) that have accessible oil and gas spill data and that are underlain by a number of shale basins (USEIA, 2011). Each of the states experienced an increase in horizontal hydraulic fracturing over the past decade (Fig. S1); however, they vary in production type from ND being primarily an oil producer to PA being primarily a gas producer. These states provide a representative range in information on spills related to UOG, which should be applicable to many geographic and ecological settings. Each state's laws and reporting requirements also vary for when (volume threshold or potential impact to people or water) and how spills are reported (verbal or written and reporting requirements) (CDPHE, 2009, NM Admin Code R. 19.15.29, ND Admin. Code R. 43-02-03-30, 25 Pa. Code § 78.66). These differences drive the quantity and quality of spills data reported in each state; therefore, a higher spill rate in a state may just reflect a more robust reporting system.

There is also considerable variability in stream setback regulations for UOG across the states examined in this study. In CO, new oil and gas development must generally avoid “restricted surface occupancy areas”, which are defined to include areas within 300 ft (91.4 m) of the ordinary high water mark of any stream segment located within designated Cutthroat Trout habitat, or streams or lakes designated by the Colorado Parks and Wildlife as “Gold Medal” (2 Colo. Code Regs. 404-1:100, 404-1:1205). Operators proposing to drill a well on a previously undisturbed site must first obtain a location assessment from the Colorado Oil and Gas Conservation Commission and must indicate on the form provided for this assessment that the proposed site is within a restricted surface occupancy area (2 Colo. Code Regs. 404-1:303b(3)(P)). Further, no new well may be drilled within 300 ft (91.4 m) of a public water system without a variance and consultation with the CO Department of Public Health and the Environment (2 Colo. Code Regs. 404-1:317B(c), and a well may only be located between 301 and 2640 ft (91.7 and 804.7 m) from a public water system if additional security measures are taken and baseline water quality testing is conducted (2 Colo. Code Regs. 404–1:317B(d)(4), (e)(2)).

In NM, wellhead protection areas include 200 ft (61.0 m) of a private, domestic freshwater well or spring used by less than five households or within 1000 ft (304.8 m) of any other freshwater well or spring (NM Admin Code R. 19.15.2.7(W)(8)). Further, permanent waste pits (NM Admin. Code R. 19.15.17.10(A)(5)) and recycling containments (NM Admin. Code R. 19.15.34.11) are not to be located within 300 ft (91.4 m) from a continuously flowing watercourse or 200 ft (61.0 m) from any other significant watercourse or lakebed, sinkhole or playa lake. Operators may obtain a waiver from these requirements if the state determines that waters will be protected from the permanent waste pit. Permanent waste pits and recycling containments also may not be located within 500 ft (152.4 m) of a spring or freshwater well, 500 ft (152.4 m) of a wetland, or within the 100 year floodplain.

In ND, Admin. Code R. 43-02-03-19 states that “[w]ell sites and facilities shall not be located in or hazardously near, bodies of water, nor shall they block natural drainage”. However, no setback distances are specified.

In PA, the edge of the disturbed area associated with an UOG well previously had to be set back 100 ft (30.5 m) or the vertical portion of the well must be 300 ft (91.4 m) from the edge of any “solid blue lined stream, spring or body of water as identified on the most current 7 1/2 minute topographic quadrangle map of the United States Geological Survey.” (58 Pa. Stat. § 3215(b)). However, this setback requirement was enjoined in *Robinson Township v. Commonwealth*, 83 A.3d 901, 1000 (Pa. 2013) and is not in effect. Pennsylvania's Act 13 also allowed the Department of Environmental Protection to require that hazardous chemicals and materials used in UOG be stored 750 ft (229 m) from blue lined streams (58 Pa. Stat. § 3215(d.1)). Finally, the Commonwealth restricts well site placement within floodplains (58 Pa. Stat. § 3215(f)).

2.2. Spill data and rates

We analyzed spill data from 01 January 2005 through 31 December 2014 for NM, ND, and PA, and from 01 January 2005 through 31 December 2013 for CO because of a significant change in CO spill reporting in 2014 (COGCC, 2014) a time period that encompassed the majority of horizontal UOG well development in these states (Fig. S1). Colorado spill data were obtained on 2 September 2014 from the state's online incident report Form 19/19A (http://cogcc.state.co.us/data.html#/cogis). Spill data within NM were accessed on 25 June 2015 from the New Mexico Oil Conservation Division website on spills (https://wwwapps.emnrd.state.nm.us/ocd/ocdpermitting/Data/Incidents/Spills.aspx). North Dakota spill data were collated on 26 August 2015 from the oilfield environmental incident reports on the North Dakota Department of Health – Environmental health website (https://www.ndhealth.gov/EHS/Spills/). Pennsylvania does not have a spill database; instead spills are reported in a Notice of Violation (NOV) database (http://www.depreportingservices.state.pa.us/ReportServer/Pages/ReportViewer.aspx?/Oil_Gas/OG_Compliance). We downloaded all PA NOVs from this site on 03 April 2015 and using the “violation codes” listed classified each violation as a “spills, potential spills” similar to Rahm et al. (2015, Table S1).

Each spill or NOV record was individually examined for duplication, errors, and a description of material and volume spilled. We grouped materials into ten main categories: chemicals, condensate, crude oil, diesel fuel, drilling waste, freshwater, hydraulic fracturing solution, natural gas, sediment, and wastewater. Materials not falling into these ten categories were grouped under “other”. For consistency across states, we had to group some of the reported material classifications into these more general categories. Drilling mud, cuttings, and drilling fluid were not consistently reported across states so these were grouped together as “Drilling waste”; similarly brine, flowback, and produced water were grouped as “Wastewater”. Hydraulic fracturing solution and frac fluid were reclassified as “HF Solution”, and chemicals used in the development process (e.g., HCl, antifreeze, surfactant, and glycol) were classified as “Chemicals”. For some records more than one material was reported; for these we treated each material as a separate spill. In ND, 20 records had no material spilled and were removed from analysis. North Dakota also had three records with a zero value for volume spilled; we classified these records as no volume reported.

We obtained well information from 01 January 1995 through 31 December 2014 from the IHS Enerdeq database (IHS, 2016), a private source for well information that synthesizes and independently quality assures data from state agencies and organizes it in a user-friendly searchable format. We used this single, private source for well information to minimize data inconsistencies among states. Because there was no clear designation of what constituted an UOG well in this database, we used geologic province name, play type, well status, and well bore orientation to identify UOG wells. UOG geologic provinces included the Denver Basin, Green River Basin and Piceance Basin in CO; Williston Basin for ND; Permian Basin and San Juan Basin for NM and Appalachian Basin for PA. We therefore considered wells as UOG if they 1) overlapped with the mentioned geological provinces, 2) had a play type classified as Shale Gas, Tight Gas, or Tight Oil, 3) had a well status of oil, gas, abandoned, pilot or suspended designation, and 4) had a horizontal hole direction.

We merged the entire spills dataset to the UOG well dataset using the American Petroleum Institute (API) number, a unique number for all oil and gas wells drilled in the U.S. This resulted in a dataset that contained only those spills associated with horizontally drilled UOG wells since 1995. We calculated spill rates by material for each state using the number of spills for a given year divided by the cumulative number of spudded wells since 1995. We used the cumulative number of wells because we wanted to examine the risk of spills throughout the life of a well. We multiplied the resulting rates by 100 thus report spill rates per 100 wells.

2.3. Potential risks to people and nature

Because streams support numerous freshwater taxa, including those for human consumption, and provide a source of drinking water for humans we assessed the potential risks of spills to these ecosystems two ways. First, we compared distances of spills to the nearest stream among states by measuring proximity (linear geographic distance) of spills to a stream using the NHDplusV2 high resolution flowline dataset (McKay et al., 2012) and the Near Tool in ArcGIS 10.2.2 (ESRI, Redlands, California, USA). We assumed that spills occurring closer to streams can pose a higher risk than spills occurring further away. Geospatial information of spills was not available in the state spill reports; therefore we used the latitude and longitude of wells as the spatial location of the spill. We acknowledge that some spills associated with supporting infrastructure and equipment of the well, e.g., storage tanks and flowlines, may not be located at close proximity to the well head, however locational data for such structures were not available for this analysis. We therefore used the well head coordinates as a surrogate, potentially over- or under-estimating actual distance of spills to streams; however, we have no reason to believe there would be a systematic bias for over- or under-estimation of distances. Second, using the distance to stream data, we calculated the number of spills in each state that occurred within the various state setback distances from streams listed above (30.5 m, 61.0 m, 91.4 m, 152.4 m and 228.6 m).

However, the importance of streams as sources of drinking water varies greatly across the U.S. Therefore, we also explored risks to drinking water among states using the U.S. Forest Service's Forest to Faucets data set (Weidner and Todd, 2011). Based on water production and water use, the Forest to Faucets data produced a Hydrologic Unit Codes (HUC) 12 watershed level index of relative importance to surface drinking water, which ranks relative importance from least important, 0, to most important, 100. We used this importance index to examine potential risk of UOG spills to surface drinking waters. All data analyses and figures were performed in R (R Development Core Team, 2015) or ArcGIS 10.2.2.

3. Results

3.1. Well and spill temporal patterns

We identified 6622 UOG spills from 5958 unique reports in the four states' databases (Table 1). Six hundred and fifty eight reports (11.1% of total) had more than one material reported on the same incident report. North Dakota had the most horizontal UOG wells, followed by PA, NM, and then CO (Table 1). While all states showed a sharp increase in the number of horizontal wells over time (Fig. S1), the number of spills increased sharply in the late-2000s only for ND and PA; for PA, the increase was followed by a decreasing trend (Fig. 1A). The maximum number of reported spills in one year was 78 in 2013 for CO, 170 in 2014 for NM, 1374 in 2014 for ND, and 324 spills in 2010 for PA (Fig. 1A, Table S2). Spill rate increased for all states except PA where the rate initially increased, reached a maximum of 20.3 spills per 100 wells in 2009 and then decreased (Fig. 1B, Table S2).

3.2. Spilled materials and volumes

We were able to identify materials for 6082 spills (91.8% of all spills). The predominant material spilled varied across states, with wastewater

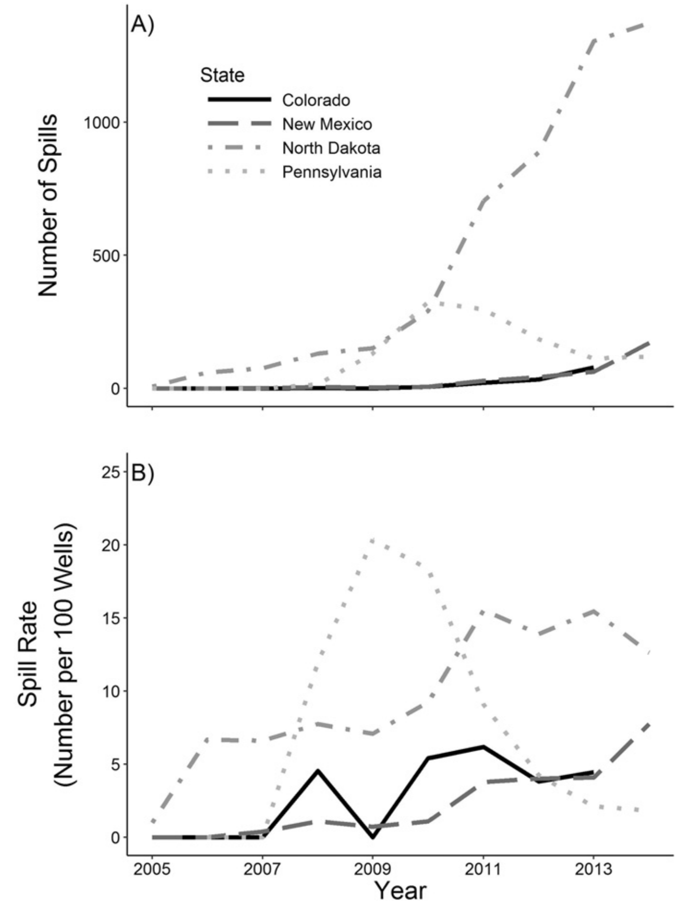

Fig. 1. Number of identified unconventional oil and gas spills (A) and number of spills per 100 wells spudded cumulatively since 1995 (B) in Colorado, New Mexico, North Dakota and Pennsylvania from 01 January 2005 through 31 December 2014. Note: numbers do not denote unique spills because we considered spill reports with more than one material as separate spills. Data for both graphs are located in Table S2.

in the top three spilled materials for all states (excluding unknowns for PA) and crude oil in the top two spilled materials for CO, NM and ND (Fig. 2A, Table S3). Top spilled material rates per 100 spudded wells consisted of crude oil (2.7% of wells), drilling waste (2.5%), wastewater (1.7%) and HF solution (0.5%) in CO; wastewater (6.3%), crude oil (5.5%), natural gas (1.3%) and drilling waste (0.5%) in NM; crude oil (24.1%), wastewater (14.1%), drilling waste (3.2%) and HF solution (1.4%) in ND; and unknown (6.4%), drilling waste (3.6%), wastewater (3.3%) and natural gas (1.9%) in PA (Table S3, Fig. 2B). A total of 5466 spills (82.5%) had reported volumes ranging from 0.4 to 3,752,100 L (Table S4; mean 7119 L, median 757 L). Volumes of the four most frequently spill materials (wastewater, crude oil, drilling waste, and HF fluid) largely ranged from 100 to 10,000 L (Fig. 3). Except for condensate, the largest-volume spills of a specified material all occurred in ND (Fig. 3, Table S4). While freshwater spills were uncommon ($n = 57$

Table 1
Number of wells and spills by state. "with volumes" column indicates the number of spills with volume data; "with materials" column shows the number of spills with materials spilled data. Numbers in parentheses indicate percentage of the number of wells (unique records, total) or percentage of total spills (with volumes, with materials). The unique records column indicates the number of unique spill reports; the Total column does not show total unique spills because we considered spill reports with more than one material as separate spills.

State	Number of wells	Number of spills			
		Total	Unique spill records	With volumes	With materials
Colorado	1753	139 (7.9)	125 (7.1)	135 (97.1)	136 (97.8)
New Mexico	2197	316 (14.4)	265 (12.1)	276 (87.3)	312 (98.7)
North Dakota	10,881	4986 (45.8)	4428 (40.7)	4859 (97.5)	4868 (97.6)
Pennsylvania	6469	1181 (18.3)	1140 (17.6)	196 (16.6)	766 (64.9)
Totals	21,300	6622 (31.1)	5958 (28)	5466 (82.5)	6082 (91.8)

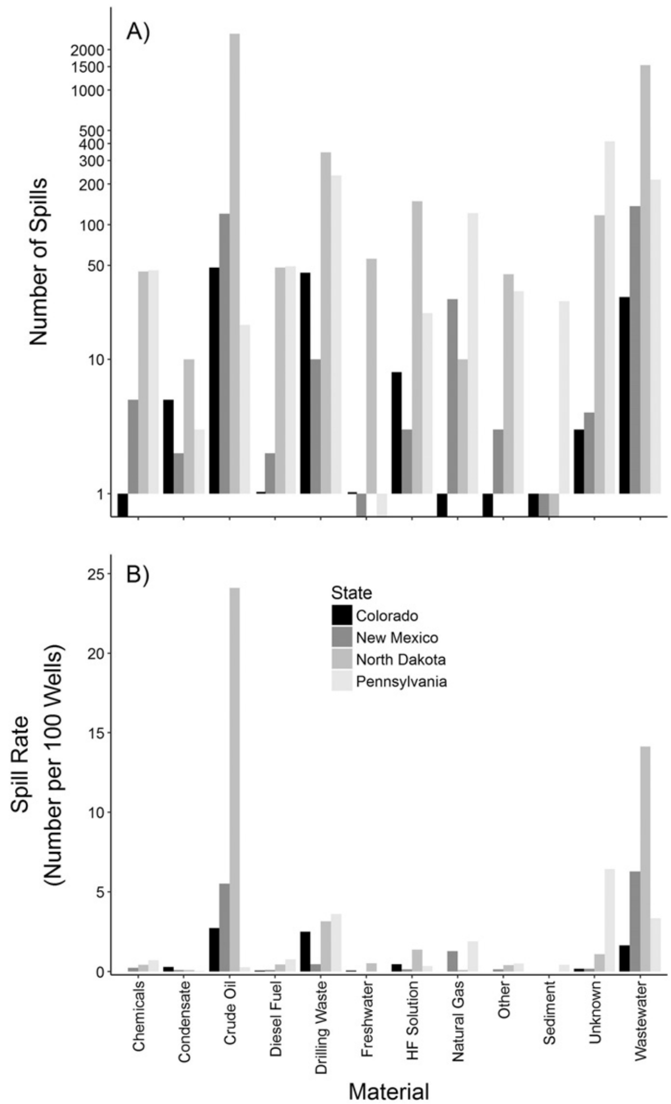

Fig. 2. Number of spills by material for each state (A) and number of spills by material per 100 wells spudded cumulatively since 1995 (B). "Other" includes all materials not included in the 10 listed, and "Unknown" indicates no material was listed in the spill reports. We note that data on the y-axis in panel A are presented on a log scale for display purposes. In panel A, bars below 1 line indicate no spills. Data for both graphs are located in Table S3.

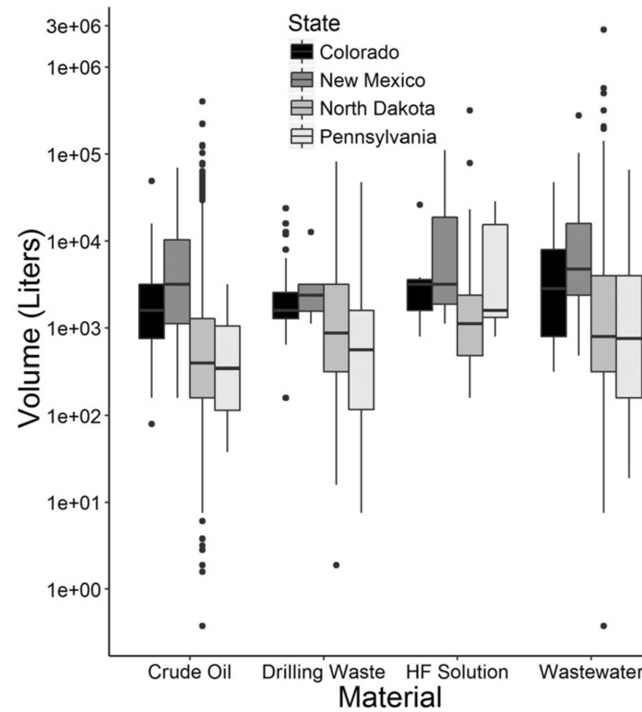

Fig. 3. Volume (L) of spills for the four most often reported spilled materials by state. Box lower and upper hinges correspond to the 1st and 3rd quartiles (25th and 75th percentiles), solid horizontal line are medians, lower whiskers extend from the hinge to the lowest values within 1.5 times the interquartile range (difference between first and third quartiles) of the hinge, the upper whiskers extend from the hinge to the highest value that is within 1.5 times the interquartile range of the hinge, and points beyond whiskers are outliers. We note that data on the y-axis are presented on a log scale for display purposes. Data for graph are located in Table S4.

total, 56 in ND, Table S3), their median volume (7949 L) exceeded the median volume of all other spills (715 L) by >10 fold.

3.3. Potential risk to people and nature

Across all states the average distance from a spill to a stream was 580 m and ranged from a low of 0.4 m in PA to a high of 9276 m in NM (Table S5). By state, average distance of spills (from wellhead) to the nearest stream was highest in NM (1379 m), followed by CO (747 m), ND (598 m) and then PA (268 m) (Fig. 4, Table S5). Across all states, 7% of total spills were within 30.5 m (100 ft setback) of a stream, 13.3% were within 61.0 m (200 ft setback), 20.4% were within 91.4 m (300 ft setback), and 47.1% were within 228.6 m (750 ft setback) (Table 2). Pennsylvania has the smallest setback regulation at 30.5 m and 5.3% of the spills in this state were within this distance to a stream. Colorado, NM and PA all had setbacks of 91.4 m and 11.5%, 9.8% and 17.4% of spills in these states, respectively, occurred within this distance from a stream. North Dakota had no specified setbacks from streams, and other than PA at the 228.6 m distance, had the highest percentage of spills within each setback distance (Table 2).

The overall average index of watershed importance to surface drinking water was highest in PA; PA values were on average 3.2 times higher than CO, 7.5 times higher than NM and 3.8 times higher than ND (Fig. 5A, Table S6). Concurrently, UOG spills also occurred in watersheds with highest importance to surface drinking water in PA (85.1), followed by CO (23.9) and ND (23.0), and lowest in NM (5.8) (Fig. 5B).

4. Discussion

Understanding the characteristics of spills is key to effectively evaluate environmental risk due to UOG development. Risk is a function of the frequency of spills, the type of material spilled, the volume of material spilled, and the proximity of the spill to surface waters and other ecologically sensitive systems. In our study, wastewater and crude oil were two of the most frequently spilled materials across all states, which is consistent with previous reports (COGCC, 2014; USEPA, 2015a). Further, a large subset of spills occurred within current setback regulation distances, which suggest some risk to streams is occurring within these distances. Spills also occurred in watersheds highly important to surface drinking water especially in PA, inferring this state's freshwater resources for drinking may be at higher risk.

The prevalence of wastewater and crude oil spills is likely a result of the large amount of both materials produced, stored and transported (Maloney and Yoxtheimer, 2012; Kondash and Vengosh, 2015). Wastewater is often high in salinity, toxic trace elements, naturally occurring radioactive materials, and other constituents depending on the producing formation and fluid mix involved in fracturing (Rowan et al., 2014; Lauer et al., 2016). Exposure to wastewater has been shown to increase plant mortality of terrestrial plants (Adams, 2011), reduce juvenile mussel survival rates (Patnode et al., 2015), and lower streambed

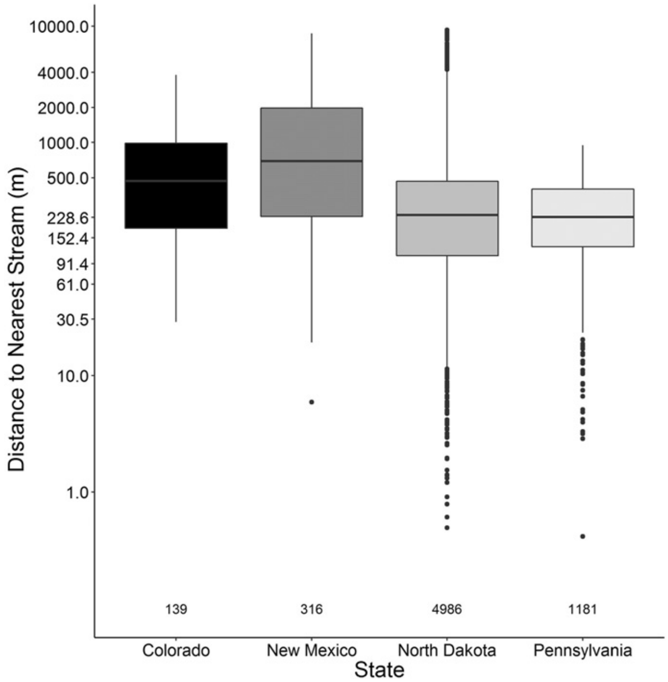

Fig. 4. Distance of spills to the nearest stream (NHDplus high resolution flowline) by state. Numbers below boxplots signify sample size. Box lower and upper hinges correspond to the 1st and 3rd quartiles (25th and 75th percentiles), solid horizontal line are medians, lower whiskers extend from the hinge to the lowest values within 1.5 times the interquartile range (difference between first and third quartiles) of the hinge, the upper whiskers extend from the hinge to the highest value that is within 1.5 times the interquartile range of the hinge, and points beyond whiskers are outliers. We note that data on the y-axis are presented on a log scale for display purposes, and we included common setback distances discussed in text. Data for graph are located in Table S5.

microbial diversity (Akob et al., 2016). Modeling and field studies suggest chemicals associated with wastewater spills can persist in the environment for several years (Rogers et al., 2015; Lauer et al., 2016). Similarly, crude oil spills can have long-term environmental impacts due to the tendency of waterway sediments to adsorb and retain oil's hydrophobic constituents (NASEM, 2016). Besides direct mortality and alterations of community structures, crude oil toxins can negatively affect individuals by causing genetic damage, enzymatic and hormonal changes, immuno-suppression, and bioenergetics and behavioral alterations (NASEM, 2016; Perhar and Arhonditsis, 2014).

Other materials that were spilled less frequently, including hydraulic fracturing fluids (HF solution) and drilling waste, may also pose environmental risks. Nearly 1000 chemicals have been used in various HF solution formulations (Konschnik and Dayalu, 2016); only a small subset of chemicals are used for each individual well and the sum of these additive ingredients is frequently only 1–2% by mass of the total HF solution (USEPA, 2015b). Nevertheless, the large volume of HF solution used in the fracturing process (Kondash and Vengosh, 2015) increases the potential for release of these chemicals to the environment. Moreover, many of the chemicals most frequently used are hydrocarbons such as light petroleum distillates (Konschnik and Dayalu, 2016), which can persist in the environment. Drilling waste, a composite of drill cutting, mud, and fluids, was mostly reported in ND and PA. These wastes, especially the cuttings, could contain low levels of naturally-occurring radioactive materials and other constituents, such as trace metals (Johnson and Graney, 2015). Finally, it is interesting to note that while not necessarily as frequent, freshwater spills were on average much larger than other spills, suggesting that frequency alone may underestimate the potential effects of spills. Potential ecological effects from a freshwater spill differ from the other materials and could include increased erosion and sedimentation.

This is the first report we are aware of that reports on proximity of spills to streams. Spills in all states occurred within their smallest setback distances from streams (30.5 m in PA; 61.0 m in NM; and 91.4 m in CO). It is possible that the close proximity of these spills to streams was the result of using a dataset that contained intermittent and ephemeral streams. However, our goal was to identify any possible conduit to a stream, even if a non-flowing system, hence the close proximity to spills indicates a potential for spilled material to reach larger streams. The number of spills within setbacks also could be because we identified the spatial position of a spill using the coordinates of the wellhead rather the supporting equipment such as flowlines or storage tanks, however as indicated before, we have no reason to believe a directional bias in over- or under-estimating spill distances. Finally, the number of spills within setbacks could be a result of waivers in some state codes (e.g., 58 Pa. Stat. § 3215(b)) that allow construction of wells within the setback; however data to evaluate this were not available for our study; a future analysis could focus on wells that were granted waivers or exemptions.

We recognize that other factors, in addition to proximity, influence the vulnerability of streams to spills. Spilled fluids may reach streams by overland flow or following infiltration and flow along subsurface pathways. The vulnerability of streams to spills that move via the subsurface track will depend, in part, on proximity, but will also reflect hydrogeologic properties that govern water-table depth, the rate and directions of groundwater flow, and whether the stream gains water from, or loses water to, the groundwater reservoir. The site-specific nature of these properties implies that assessment of threats that spills pose to stream-water quality must be conducted on a case-by-case basis.

Ecosystems provide many benefits to people, including food and water, regulating floods and diseases, and recreation (Millennium Ecosystem Assessment, 2005). We evaluated the potential effects of UOG spills on ecosystem services by focusing on surface drinking water, using an importance index developed by the US Forest Service. We found that spills in Pennsylvania occurred in watersheds with a much higher value to surface water than the other states, a result of higher population density and reliance on surface waters as drinking water in this area (Weidner and Todd, 2011). The other three states had spills in watersheds with lower importance values, which likely reflects a greater reliance on groundwater or that drinking water is sourced from streams and reservoirs of distal watersheds. Nevertheless, in these states a localized spill that affects water reservoirs and treatment plants can negatively impact communities near UOG development.

Several approaches to model the risk of UOG development have been recently implemented, from probability modeling (Rozell and Reaven, 2012) to expert risk assessments (Walker et al., 2016). Our results suggest that modeling the risk of hydraulic fracturing will require region-specific parameterizations. For example, North Dakota had the highest spill rates for both crude oil and wastewater as well as the largest reported spill volumes for these materials. The Bakken is the largest producer of crude oil (USEIA, 2016) and wastewater (Kondash and Vengosh, 2015) for plays in our study, which could be one reason for

Table 2
Number of spills within specified distance of streams corresponding to existing setback regulations. Numbers in parentheses indicate the percentage of spills within the setback distance out of total spills (see total column in Table 1). Grey shading represent setback distances specified for each state depending on regulation, where North Dakota had no setback distances specified.

	Distance (m)				
State	30.5	61.0	91.4	152.4	228.6
Colorado	2 (1.4)	7 (5.0)	16 (11.5)	28 (20.1)	41 (29.5)
New Mexico	13 (4.1)	25 (7.9)	31 (9.8)	47 (14.9)	78 (24.7)
North Dakota	388 (7.8)	736 (14.8)	1095 (22.0)	1722 (34.5)	2410 (48.3)
Pennsylvania	63 (5.3)	116 (9.8)	206 (17.4)	378 (32.0)	589 (49.9)
Totals	466 (7.0)	884 (13.3)	1348 (20.4)	2175 (32.8)	3118 (47.1)

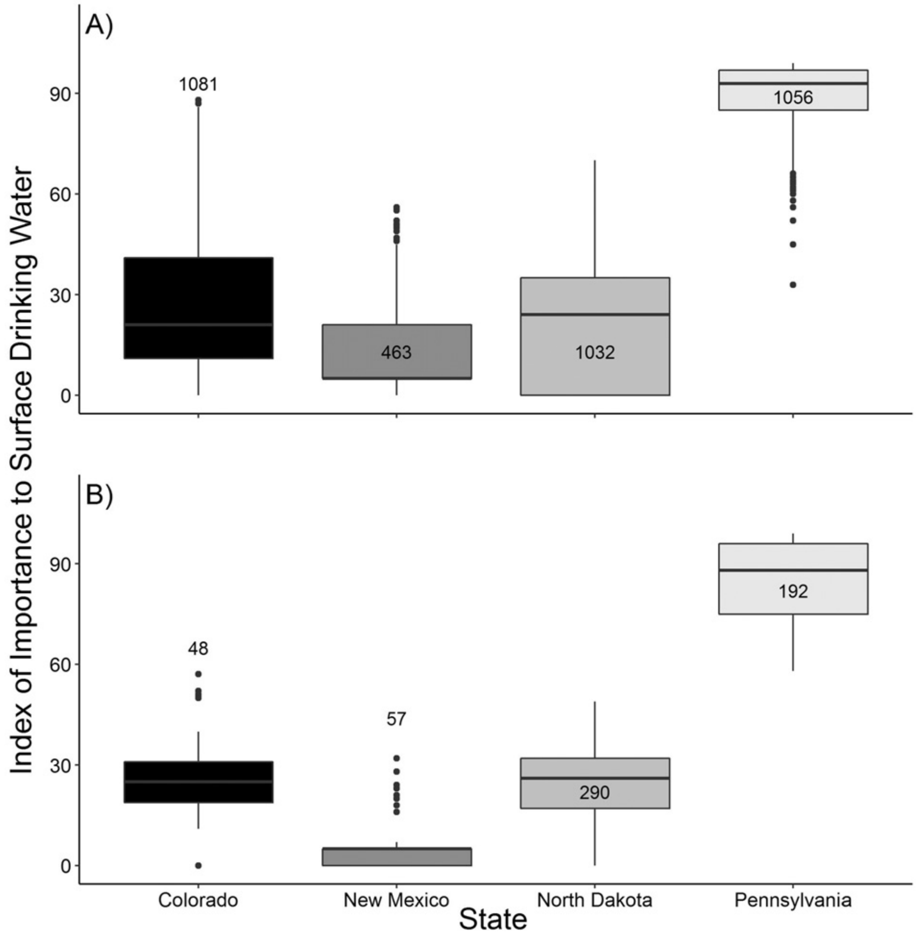

Fig. 5. Forest to Faucet index of importance to surface drinking water for all HUC12 watersheds in the Forest to Faucet data set in each state (A) and for the subset of these HUC12 watersheds in which a UOG spill occurred (B). Number within or above boxplots are sample size. Box lower and upper hinges correspond to the 1st and 3rd quartiles (25th and 75th percentiles), solid horizontal line are medians, lower whiskers extend from the hinge to the lowest values within 1.5 times the interquartile range (difference between first and third quartiles) of the hinge, the upper whiskers extend from the hinge to the highest value that is within 1.5 times the interquartile range of the hinge, and points beyond whiskers are outliers. Data for graph are located in Table S6.

the higher spill rates. Another reason may be development rate, where ND had over twice the number of spudded horizontal wells in 2012, 2013 and 2014 than the other three states. Finally, differences in reporting are also a factor; ND requires reporting at a lower threshold (1 barrel; 159 L) than either CO or NM (5 bbls, 795 L), and its dataset also had the most complete records in terms of materials and volumes associated with spills. Unifying reporting requirements across states would aid future broad-scale data analyses and risk modeling.

Finally, we acknowledge that all spills may not have been reported or included in the available state databases. Data were not available to estimate the frequency of these "missing" spills and we are unaware of any study that has directly examined missing spills (but see Wiseman, 2013). To deter under-reporting, states in our study have penalties for failing to report a required spill that include maximum civil penalties that range across the states from $12,500 (ND) to $15,000 (CO) per day to $75,000 plus $5000 per day (PA); NM, ND, and PA also categorize a failure to report as a possible criminal act (Colorado House Bill 14–1356, Section 1, 34–60-121; North Dakota Century Code Section 38–08-16.1; New Mexico House Bill 286 Section 70–2-31(B); 58 Pa State § 3255, 3256). Further, our large sample size (n = 6622) from four states over ten years likely averages any effects of under- or over-reporting, and missing spills are thus not likely to affect overall patterns.

5. Conclusions

Concerns over potential environmental issues resulting from UOG development have spurred a flurry of recent articles on its potential ecological effects to associated ecosystems (Brittingham et al., 2014; Evans and Kiesecker, 2014; Souther et al., 2014). An important gap in our understanding of potential effects of UOG on associated ecosystems is the surface release of chemicals, waste materials and oil and gas. We found that wastewater, crude oil, drilling waste and hydraulic fracturing solution are the materials most often involved in a spill incident, and that most spills ranged between 100 L and 10,000 L. We also found that some spills occurred in very close proximity to streams and in watersheds of high importance to drinking water. Further, spills occurred within current setback requirements. As UOG activity continues in the U.S. and commences in other countries around the world, our findings can be used to better understand the risk of spills from UOG development and help inform management decisions, policy, and regulations.

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Competing financial interest declaration

The authors declare no competing financial interests.

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ATTACHMENT C

STUDY 39

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Article

Soil Erosion and Surface Water Quality Impacts of Natural Gas Development in East Texas, USA

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Abstract: Due to greater demands for hydrocarbons and improvements in drilling technology, development of oil and natural gas in some regions of the United States has increased dramatically. A 1.4 ha natural gas well pad was constructed in an intermittent stream channel at the Alto Experimental Watersheds in East Texas, USA (F1), while another 1.1 ha well pad was offset about 15 m from a nearby intermittent stream (F2). V-notch weirs were constructed downstream of these well pads and stream sedimentation and water quality was measured. For the 2009 water year, about 11.76 cm, or almost 222% more runoff resulted from F1 than F2. Sediment yield was significantly greater at F1, with 13,972 kg ha^{-1} yr^{-1} *versus* 714 kg $ha^{-1}yr^{-1}$ at F2 on a per unit area disturbance basis for the 2009 water year. These losses were greater than was observed following forest clearcutting with best management practices (111–224 kg ha^{-1}). Significantly greater nitrogen and phosphorus losses were measured at F1 than F2. While oil and gas development can degrade surface water quality, appropriate conservation practices like retaining streamside buffers can mitigate these impacts.

Keywords: water quality; surface runoff; oil and natural gas development; fracking; sedimentation; erosion; APEX model; best management practices; riparian buffers

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1. Introduction

Recent advances in drilling technology have resulted in a dramatic expansion in exploration for and development of oil and natural gas. Historically, single vertical wells were drilled into hydrocarbon traps in permeable rock formations where gas and oil had migrated to. Starting in the 1940s, water, sand, and other additives under high pressure were used to fracture low permeability hydrocarbon source rocks like shales. Due to the high cost of these operations relative to the value of the oil and gas recovered, this practice had only limited applicability. Recent advances in horizontal drilling technology coupled with higher prices for oil and natural gas have resulted in a significant increase in hydraulic fracturing or fracking. In addition, CO_2 emissions from natural gas combustion are 30%–40% lower than coal, NOx emissions are 80% lower for natural gas, and emissions are almost 100% lower for SO_2, particulates, and mercury compared with coal [1]. Therefore, natural gas is seen as an acceptable bridge fuel until more sustainable energy sources become viable. This will likely result in greater development of natural gas resources in the future.

One area of very active drilling in the United States is East Texas, southwestern Arkansas, and western Louisiana. The Haynesville, Cotton Valley, Travis Peak, and other formations underlie this region and have been very productive, with a drilling success rate of over 99%. The Haynesville shale has been the most productive formation and is between 3.1 and 4.3 km deep and about 91 m in thickness [2]. It is estimated to contain about 7 trillion m^3 of natural gas [3]. Drilling increased by over 300% in the Haynesville region from 2008 to 2012.

There are numerous concerns associated with oil and gas development and water resources. These include firstly, the large amount of water used in fracking. In the Barnett shale, fracking water use in 2010 was 308 Mm^3, or about 9% of the total water used by the city of Dallas, Texas [4]. In addition, concerns exist about the possibility of fracking fluids contaminating aquifers. With regards to surface waters, leaking pipelines, reserve pits, and producer water spills are a significant hazard [5]. Finally, concerns exist about the erosion and sedimentation that can result from natural gas development. Sedimentation is among the greatest contributors to stream impairment in the United States [6].

In the Barnett shale region of north Texas, sediment yields from natural gas sites in Denton County were 54 t ha^{-1} yr^{-1}, much greater than the 1.1 t ha^{-1} yr^{-1} measured from undisturbed rangelands in this region [7]. The United States Environmental Protection Agency (USEPA) regulates small construction sites (0.4 ha or greater) for stormwater discharge and sediment movement. In the state of Texas, gas wells are not regulated by the state environmental agency as small construction sites and are not subject to the same regulations. In addition, little regulatory oversight is given to how the placement of well pads may impact surface water resources.

Best management practices (BMPs) to control stormwater discharge and nonpoint pollution for other industries like agriculture and forestry have been widely adopted in the USA. For example, over 95% of forestry operations in Texas employ these BMPs [8], and these BMPs have been proven to be very effective in reducing sedimentation from clearcutting and site-preparation [9]. Similarly, it is estimated that sedimentation from natural gas well sites could be reduced by as much as 93% by using BMPs [10].

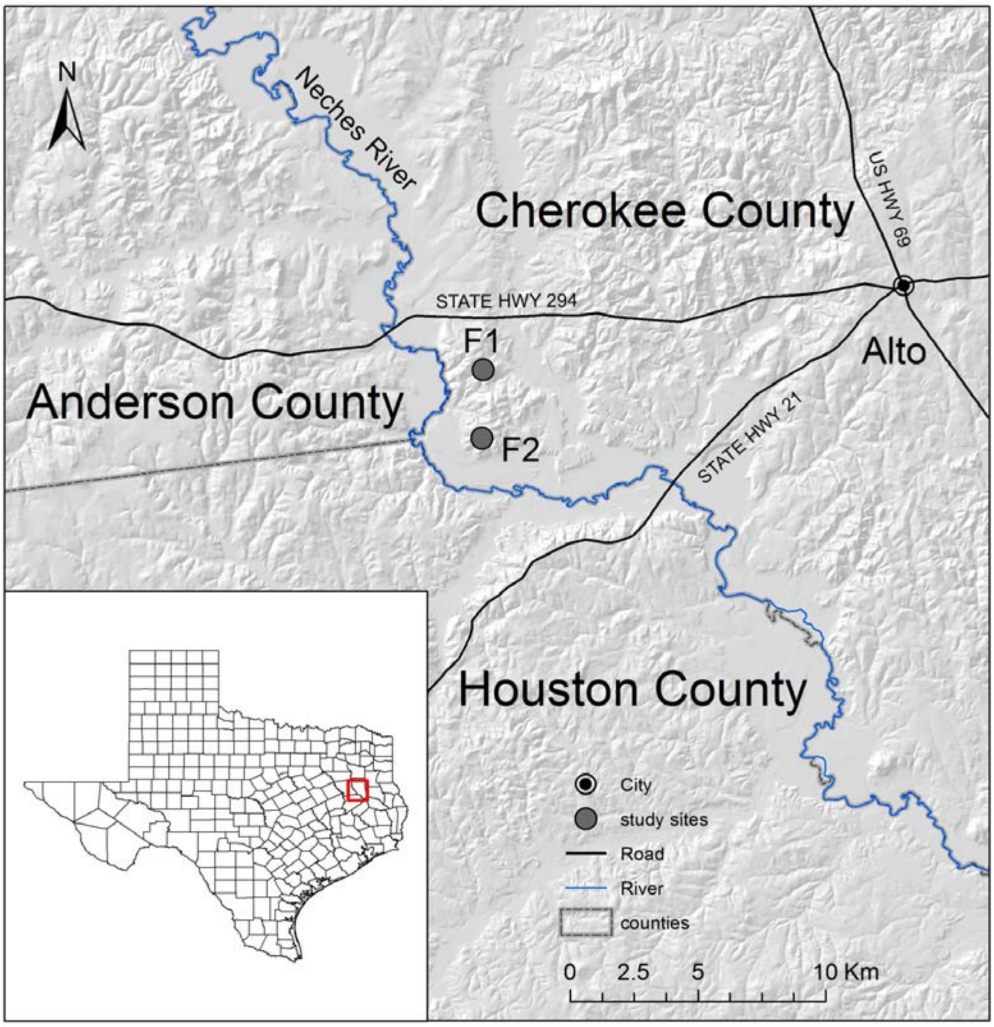
The purpose of this study was to quantify the stormwater concentrations and losses of sediment, nutrients, and metals from a natural gas well site. Comparisons were made between a gas well site constructed in the stream channel and a site offset from the stream channel by 15 m to determine the extent to which well location may affect sediment loss and water quality. Comparisons were also made between these water quality impacts and impacts from other land uses in the watersheds.

2. Materials and Methods

2.1. Study Area

The study was conducted at the Alto Experimental Watersheds in the Neches River basin approximately 16 km west of the town of Alto in Cherokee County, Texas, USA (Figure 1). The study area is in the Gulf Coastal Plain and has a humid subtropical climate. Average summer temperatures are 27.2 °C and average winter temperatures are 9.5 °C, with a mean annual temperature of 18.7 °C. Annual rainfall in the region is 117 cm. The rain is distributed fairly evenly throughout the year with an average of 89 rain days a year, with April and May receiving the largest amount of rainfall [11].

Figure 1. Location of study watersheds (F1 = no riparian buffer, F2 = 15 m riparian buffer) at the Alto Experimental Watersheds in Cherokee County, Texas, USA.



The soils at the Alto Experimental Watersheds formed in Eocene sediments. The dominant surface formations are members of the Claiborne Group and are Sparta Sand and the Cook Mountain

Formation [12]. These soils developed under mixed loblolly pine (*Pinus taeda*) and hardwood forests, have low inherent fertility and are most commonly classified as Alfisols and Ultisols. The most prevalent soil found in the watersheds is the Sacul Series (fine, mixed, active, thermic Aquic Hapludults) followed by the Tenaha Series (loamy, siliceous, semiactive, thermic Arenic Hapludults). Both soils are Ultisols with an argillic horizon and less than 35% base saturation. Teneha soils are well drained and runoff is negligible to medium with increasing slope [13]. Sacul soils are slowly permeable soils that formed in acidic, loamy and clayey marine sediments. They are moderately well drained with medium to very high runoff potential, and have a seasonally high water table that is within 61 to 122 cm of the soil surface in late winter and spring most years [13].

2.2. Treatments

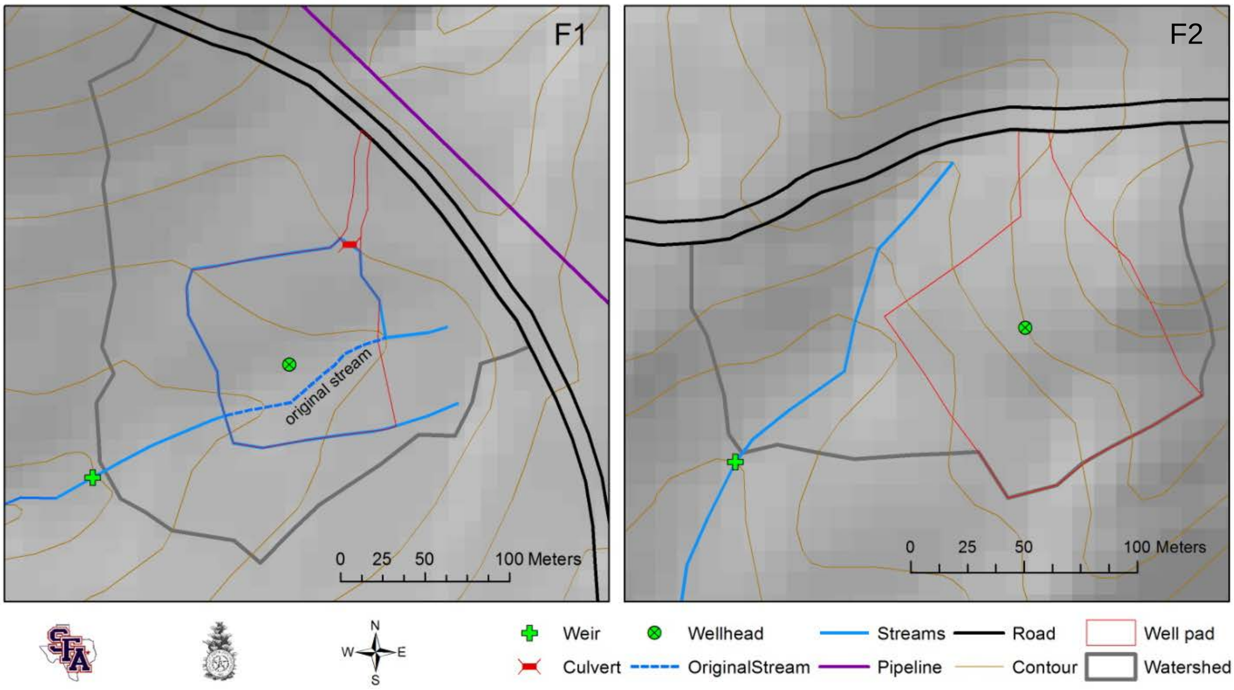
In the spring of 2008, two natural gas wells were drilled. At the first site (F1), the well pad was constructed directly in the channel of an intermittent stream and has a watershed area of 13.7 ha with the pad comprising 1.4 ha (Figure 2a). The stream was rechanneled around the north side of the pad following construction. At the second site (F2), the pad was offset from the creek channel by about 15 meters; this site has a watershed that consists of 4.5 ha with the well pad occupying 1.1 ha (Figure 2b).

In the process of constructing the well pad at F-1, fill material had to be brought in from an undisclosed location. The fill material consisted of 55.5% sand and 44.5% clay. Once this fill material had been brought in and the site leveled, iron ore gravel (16–150 mm diameter) was hauled in and spread over the majority of the pad with the exception of approximately one-quarter of the western end of the pad, which was used for a drilling fluid reserve pit. After drilling was completed, the reserve pit was filled with soil that was 40.2% sand, 14.1% silt, and 45.7% clay. This area was then seeded with ryegrass (*Lolium* spp.) While some of the seeds germinated, most did not grow or were carried away by surface runoff, resulting in bare soil.

The well pad at F2 required no fill material for pad construction due to the topography of the site. F2 was placed on the southern face of a large hill. Earth-moving equipment was used to modify the hill from a steep slope to a 1.1 hectare terrace suitable for operating large drilling equipment on. This soil was 65.1% sand, 9.5% silt, and 25.3% clay. After the terrace was constructed, iron ore gravel was spread similar to the method employed at F1. The back, southern portion used as a reserve pit for drilling fluids. The soil used to fill in the reserve pit was 21.7% sand, 32.1% silt, and 46.2% clay.

Both sub-watersheds where the gas well sites were constructed were dominated by loblolly pine. The northern portion of the F1 watershed was mixed hardwoods and pine; this area comprised approximately 3.5 hectares. The rest of the F1 watershed was 10–15 year old loblolly pine plantation. Approximately 2 hectares of the F2 watershed was 10–15 year old loblolly pine plantation while the rest was a mixed hardwood and pine stand. The portion of the watershed that was mixed hardwood and pine was composed of fairly large (≈50–100 cm) timber. These larger diameter trees consisted primarily of white and red oaks (*Quercus* spp.) and loblolly pine. This area of large mixed timber at both watersheds was the result of timber harvests in compliance with Texas BMPs, leaving the riparian forest as a contiguous buffer known as a streamside management zone (SMZ). The understory of both watersheds consists mostly of species such as dogwood (*Cornus florida*), sweetgum (*Liquidambar* styraciflua), various magnolias (*Magnolia* spp.), various hickories (*Carya* spp.), yaupon (*Ilex vomitoria*), sassafras (*Sassafras albidum*), and American beautyberry (*Calicarpa americana*).

Figure 2. F1 (**a**) and F2 (**b**) natural gas well pad layout at the Alto Experimental Watersheds, Texas, USA.



(**a**) (**b**)

2.3. Water Quantity and Quality

In both streams, a v-notch weir was constructed approximately 80 m downstream from the pad (Figure 3).

In each weir, an AquaRod® water level monitor was installed in the mouth of the flume. Unfortunately, stage data obtained from the AquaRods® were unreliable due the unexpectedly high sediment loads deposited in the weirs burying the capacitance rods. Streamflow was therefore estimated using the ArcAPEX model from precipitation measured at the sites [14]. ArcAPEX was calibrated and validated for these watersheds in earlier studies [15]. Rain gauges were located throughout the watershed and after each storm event precipitation data were collected.

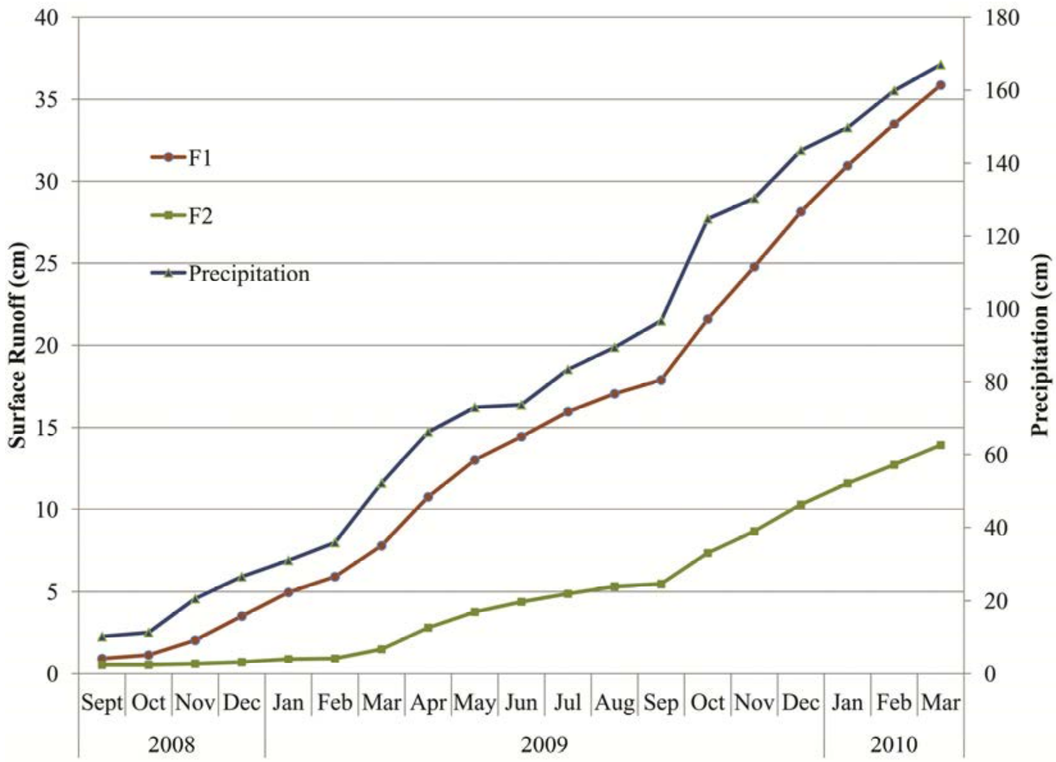
As a result of the streamflow being ponded by the front plate of the weir, the coarse sediments were deposited in the drop box section on the floor of the weir. After each rain event this sediment was removed and weighed to determine the amount of sedimentation occurring in the stream channel (Figure 4). Dry mass was determined from a sub-sample of this sediment. The amount of sediment deposited in the drop box was later added to the amount of suspended sediment losses in stormflow. These losses were quantified using the flow estimated by ArcAPEX multiplied times the total suspended sediment (TSS) values that were obtained from stormwater samples. Sampling occurred from September 2008 to March 2010.

Water samples were collected from each weir using one of two techniques. The first technique utilized a Nalgene® Storm Water Sampler (Figure 3). Within 24 h of each storm runoff event the sample bottle was removed and a clean, acid rinsed bottle was placed in the cylinder. These samplers were frequently buried by the large volumes of sediment. When this occurred, the second method was used, the grab sample method, in which a 1 L sample bottle was placed in the flow of the stream and a water sample was taken. Grab samples typically represented the recession phase of the hydrograph. Once the samples were collected from the field they were brought to the laboratory for analysis. The samples in the lab were analyzed using a Hach® DR/890 Datalogging Colorimeter and a Hach® sensION 156 Portable pH/Conductivity Meter according to approved United States Environmental Protection Agency (USEPA) methods [16]. Parameters analyzed included total suspended solids (TSS), total dissolved solids (TDS), pH, conductivity (EC), total nitrogen (TN), ammonia (NH_4^+), nitrate nitrogen (NO_3^-), nitrite nitrogen (NO_2^-), total phosphorus (TP), ortho-phosphate (PO_4^+) sulfate (SO_4^+), iron (Fe), turbidity, color, salinity, calcium hardness and magnesium hardness. A paired T-test was employed to determine if mean water quality values were different by site at = 0.05.

Figure 3. In-channel instrumentation for measuring total runoff (V-notch weir), stream level (Aquarod®), water quality (Nalgene® Stormwater Sampler), and sediment (drop box) on the F2 sub-watershed before a storm event (**a**) and after a 6.3 cm rain event in April, 2009 at F1 (**b**); at the Alto Experimental Watersheds in Texas, USA.

(**a**) (**b**)

Figure 4. Cumulative ArcApex simulated water yield and rainfall for two natural gas well locations, one placed directly in the stream channel (F1) and the other offset from the channel by a 15 m buffer (F2) at the Alto Experimental Watersheds, Texas, USA.



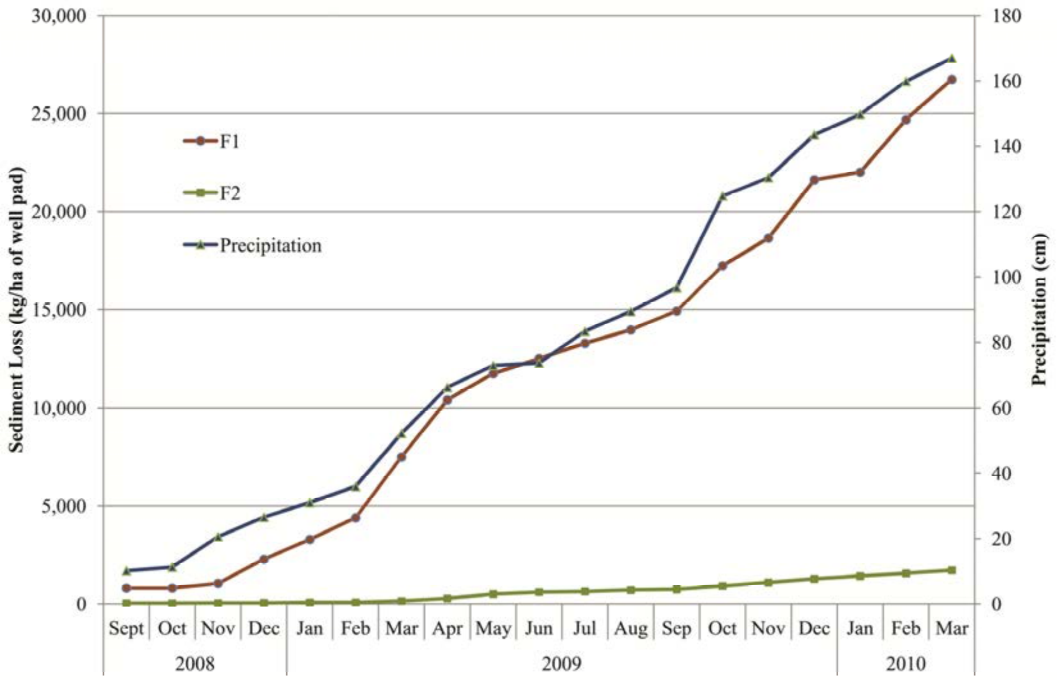
3. Results

In the small forested watersheds of East Texas, stream flow in headwater streams is typically intermittent and is mostly a product of storm runoff. The simulated water yield at F1 was significantly greater ($p < 0.0001$) than the water yield at F2 (Figure 4). In the first month of data collection (September 2008) the water yield at F1 was 0.915 cm and 0.545 cm at F2. Due to lower than average precipitation in the month of October, there was a decrease in storm runoff, but this decrease was most pronounced at F2, with 0.216 cm and 0.001 cm at F1 and F2 respectively. This trend continued throughout the study period, regardless of season. Percent runoff efficiency (runoff divided by precipitation) was different for two watersheds, 33.0% at F1 and 12.3% at F2.

Soil compaction of the well pad was much greater than in the rest of the watershed. The mean bulk density of the well pad at F1 was 2.04 g cm^{-3}. Mean bulk density measurements taken in the surrounding watershed were 1.3, 1.19, and 0.99 g cm^{-3} for logging sets, skid trails, and undisturbed forest floor respectively.

Sediment yield was also significantly greater ($p < 0.001$) from F1 that F2 (Figure 5). Starting in September 2008, the sediment yield was 83 kg ha^{-1} at F1 *versus* 10 kg ha^{-1} at F2. Continuing through the winter of 2009, the total yield continued to increase at F1 over F2. The total sediment yield for the 2009 water year (September 2008–August 2009) was 19,561 kg *versus* 785 kg at the F1 and F2 watersheds, respectively. However, this does not take into account the differences in the percent of the watershed that was actually disturbed by the well site. The well site occupied about 24% of the total watershed area at F2 *versus* about 10% at F1. Therefore, it is also useful to compare the sediment yields per unit area disturbed by natural gas development in order to make meaningful comparisons with the clearcut watersheds. On this basis, the equivalent sediment losses for F1 and F2 were 13,972.1 and 714 kg $ha^{-1}yr^{-1}$ for the 2009 water year respectively, or 16,896 and 1,087 kg $ha^{-1}yr^{-1}$ for F1 and F2, respectively, annualized for the entire 19 month (September 2008–March 2010) study period. About 56% of the sediment loss recorded at F1 was deposited in the flume, with less than 44% moving in the suspended form. However, at F2, 98% of the sediment moved in the suspended form over the study period, with only 2% being deposited in the flume. Since sediment filled the flume on F1 for several runoff events, it is possible that these loss values underestimate the amount of coarse sediments actually eroded from the pad.

Figure 5. Cumulative sediment yield and rainfall for two natural gas well locations, one placed directly in the stream channel (F1) and the other offset from the channel by a 15 m buffer (F2) at the Alto Experimental Watersheds, Texas, USA.



In terms of concentrations of other water quality parameters, differences between F1 and F2 were less pronounced (Table 1). For nutrients, only PO_4^+ was significant, with the mean value being significantly greater at F2 than at F1. At F1, pH was also significantly greater, though these values were well below the Texas water quality standard minimum value of 6.0. Color was significantly greater at F1 than F2, probably associated with the higher amounts of sediment eroded from the pad at F1. However, there were no significant differences in either TSS or TDS. Salinity was significantly greater at F1 than F2, and this could have been attributed to an accidental spill of saline producer water that occurred in October 2008, but more sampling would have been required to establish this. The volume and chemical properties of this salt water was spilled was not tested. However, this spill did result in the death of several loblolly pine trees and understory vegetation down gradient of the well pad (Figure 6).

When nutrient and metal concentrations were converted to mass losses per hectare, all of the losses were greater from F1 than F2, with TDS, TN, NO_3^-, PO_4^+, SO_4^+, and Fe being significantly greater ($\alpha < 0.05$) using the *T*-test (Table 2). Since streamflow was significantly greater at F1 throughout the study period (Figure 4), it would be expected that mass losses would also be greater.

Table 1. Mean concentrations for water quality parameters measured below two natural gas well sites (F1 and F2) from October 2008–March 2010 at the Alto Experimental watersheds in East Texas, USA.

Water quality parameter	Mean[1]		*T*-test *p*-value
	F1	F2	
Total Nitrogen (TN, mg L^{-1})	2.78	2.50	0.26
Ammonia (NH_4^+, mg L^{-1})	1.55	0.57	0.27
Nitrate (NO_3^-, mg L^{-1})	2.78	0.74	0.15
Nitrite (NO_2^-, mg L^{-1})	0.02	0.03	0.50
Total Phosphorus (TP, mg L^{-1})	0.57	0.72	0.59
Ortho-Phosphate (PO_4^+, mg L^{-1})	0.16	**<u>0.30</u>**	0.01
Total Suspended Solids (TSS, mg L^{-1})	335.72	288.33	0.40
Total Dissolved Solids (TSD, mg L^{-1})	281.43	415.44	0.13
pH	**<u>4.90</u>**	4.53	0.04
Conductivity (μS cm^{-1})	461.06	554.65	0.30
Color (CU)	**<u>1231.28</u>**	576.58	0.04
Calcium Hardness (mg L^{-1})	1.23	0.75	0.23
Magnesium Hardness (mg L^{-1})	2.81	2.95	0.87
Iron (Fe, mg L^{-1})	5.55	4.36	0.18
Salinity (mg L^{-1})	0.24	**<u>0.41</u>**	0.02
Sulfate (SO_4^+, mg L^{-1})	6.43	5.30	0.23

Note: [1] Bold underlined values were significantly greater based on the paired *t*-test at $\alpha = 0.05$.

Table 2. Total values for mass losses (kg ha^{-1}) for water quality parameters measured below two natural gas well sites (F1 and F2) from October 2008–March 2010 at the Alto Experimental Watersheds in East Texas, USA.

Water quality parameter	Sum[1]		*T*-test *p*-value
	F1	F2	
Total Nitrogen (TN)	**<u>10.84</u>**	3.08	0.00
Ammonia (NH_4^+)	4.55	0.67	0.112
Nitrate (NO_3^-)	**<u>11.84</u>**	0.84	0.035
Nitrite (NO_2^-)	0.10	0.05	0.217
Total Phosphorus (TP)	2.53	1.42	0.059
Ortho-Phosphate (PO_4^+)	**<u>0.72</u>**	0.47	0.042
Total Suspended Solids (TSS)	**<u>1,196</u>**	418	0.000
Total Dissolved Solids (TDS)	**<u>969</u>**	559	0.032
Iron (Fe)	**<u>19.2</u>**	5.54	0.001
Sulfate (SO_4^+)	**<u>21.53</u>**	6.43	0.000

Note: [1] Bold underlined total values were significantly greater based on the paired *t*-test at $\alpha = 0.05$.

Figure 6. Mortality of loblolly pine overstory trees (red/brown needles) and understory vegetation at F2 at the edge of the streamside buffer strip following an accidental spill in October 2008 of saline water produced during natural gas extraction at the Alto Experimental Watersheds in Texas, USA.

4. Discussion

4.1. Storm Runoff

Total runoff from these two natural gas well locations was much greater than would be expected from undisturbed areas in this region. In the undisturbed forested areas, direct surface runoff is uncommon. However, due to the significant increase in bare, compacted soils surface runoff was much more frequent. In addition, the significantly higher bulk density on the well locations resulted in less infiltration. McBroom *et al.* [17] found that for nearby undisturbed forests, annual runoff ranged between 0.64 and 10.32 cm, depending on rainfall. Following clearcutting of the watersheds reported by McBroom *et al.* [17], annual runoff ranged between 7.82 and 9.79 cm. This was comparable to runoff measured at F2 in the 2009 water year of 9.58 cm. However, the clearcut reported by McBroom *et al.* [17] covered an average of 75% of the total watershed area, where the well location at

F2 only occupied about 10% of the total watershed area. Even when the gas well pad was offset by 15 m from the stream, it still had a proportionally greater impact on runoff than forest management. For the well pad directly in the stream channel, the effects on runoff were much greater, with 24.67 cm of runoff in the 2009 water year. In addition, runoff efficiency following clearcutting on adjacent watersheds increased from 1% pre-harvest to 9% post harvest, compared with 33% and 12% on F1 and F2, respectively.

4.2. Sediment Losses

In terms of total sediment yield, results from this study are much greater than reported from proximate watershed studies, indicating the greater relative impact of natural gas development. For undisturbed forestlands, sediment yield averaged about 42 kg ha^{-1} [17]. Following clearcut harvesting and site preparation in 2003, losses increased from 111 to 224 kg ha^{-1} yr^{-1}, though these differences were not found to be statistically significant [17]. In that study, a streamside management zone (SMZ) with a minimum total width of 30 m was retained around all stream channels. In 1981 these same watersheds were clearcut harvested and no SMZ was retained, and the following site preparation, sediment losses averaged 2917 kg ha^{-1} first year after harvest [18]. Losses returned to levels measured in undisturbed forests by the second year after harvest in both 1981 [18] and in 2003 [15]. While large sediment plumes were observed to have eroded from both gas well locations, at F2 lobes of coarser sediments were trapped by the riparian vegetation and surface cover before reaching the stream channel. On F1, the 13,972 kg ha^{-1} of disturbance for 2009 largely resulted from sediment moving from the fill slope on the back side of the pad directly into the stream channel (Figure 7).

Construction of a natural gas well location in Denton, Texas resulted in 54,000 kg ha^{-1} yr^{-1} of erosion [7]. This represents sediment that eroded from the pad, but may not have necessarily entered the stream channel. Using the RUSLE 2.0 model, Waschal *et al.* [10] concluded that good sediment control practices and BMPs can reduce sediment yields from natural gas well pads by 52%–93%. Similarly in the current study, the 94% difference in sediment between F1 and F2 can be attributed in part to the 15 m riparian buffer on F2 and better stormwater management.

One area of continued concern on F1 is that no efforts at site stabilization or revegetation were attempted following the initial failed attempt at seeding with rye grass. Significant rill and small gully erosion resulted from storm runoff flowing off the compacted pad area and down onto the sloping fill material where the reserve pit had been. Unlike results reported by Williams *et al* [7], after four years, the F1 well pad continued to erode with little evidence of natural stabilization, and natural vegetation remained sparse due to the poor condition of this fill material as a plant growing medium.

Similar to what was found with natural gas wells in the Fernow Experimental Forest in West Virginia, silt fences were inadequate at stopping these large sediment volumes [19]. Silt fences were installed down-gradient of the well location during construction, but they were installed about 0.25 m above the old stream channel on F1 and were overwhelmed by the large sediment loads, making silt fence ineffective at controlling these large volumes of sediment (Figure 8). Like with the wells constructed in the Fernow [19], improper installation resulted in the ineffectiveness of silt fence as a stormwater BMP. Silt fences functioned as intended on F2 due to proper installation and a lower overall sediment load that did not overwhelm their design capacity.

Figure 7. Sediment plume below the F2 natural gas well location trapped by riparian forest vegetation before entering the stream channel at the Alto Experimental Watersheds in Texas, USA.

Figure 8. Silt fence installed below the natural gas well pad at F1 illustrating the ineffectiveness of this sediment control technique due to poor installation and large sediment volumes eroded at the Alto Experimental Watersheds in Texas, USA.

Beyond the continued erosion of the well pad, another significant concern that exists is that this deposited sediment will have long term consequences for the aquatic ecosystem. The original stream channel below the well on F1 was buried by about 0.5 m of sediment and the original pool, riffle, and glide aquatic habitats were obliterated. Sediment loading of this magnitude can have dramatic effects on lotic food webs [20]. For streams in the southeastern United States, hundreds of years if not millennia may be required to naturally purge large volumes of sediments out of regional stream and river networks [21]. This represents a localized legacy sediment issue comparable to what occurred in this region due to poor agricultural practices in the 19th Century. Effective and systematic implementation of soil conservation practices is needed to ensure that significant land use alterations do not impair surface waters in regions with extensive natural gas development.

4.3. Water Quality

The effects of natural gas development on water quality parameters were less significant than with sedimentation. Concentrations of most parameters were not significantly different between F2 an F1. However, overall runoff volumes were greater at F1 than F2, so when concentrations were converted to mass losses, the most of the water quality parameters were significantly greater at F1 than F2. The larger runoff volumes from F1 may have diluted the concentrations, but the overall mass export was significantly greater. This indicates that reducing the export of nutrients and metals from natural gas well pads is dependent on effective stormwater management. At F1, there was no buffering between the well pad and the stream, meaning that direct contributions of contaminants occurred without the benefits of filtration provided by riparian buffer strips.

As noted in the Results section, a producer water spill at F2 did result in the death of several trees along the stream channel, and this may account for the significantly higher salinity values at F2 (Figure 6). Differences in water quality were not observed with other parameters. Ground water was pumped into the stream channel immediately after the spill for several days in order to dilute the effects of the spill. While the water quality of the spilled water was not characterized, this remediation measure may have been adequate to reduce the impacts on water quality parameters that could be directly measured. However, the death of the riparian trees immediately in the flow area of the spill indicates that the direct ecological effects may require different remediation strategies.

5. Summary and Conclusions

Natural gas development is important for maintain economic prosperity and for providing a necessary energy source until renewable energy sources become more viable [22]. However, significant impacts on surface water resources were measured in this study when a gas well pad was constructed with little attention given to surface drainage patterns. Unfortunately, this was not an isolated incident on this lease area, with a pad being constructed in a perennial stream a few km north from F1 and another pad platted and surveyed over another intermittent stream nearby. Erosion rates that result from this practice are orders of magnitude greater than other land uses in this region. The 13,972 kg ha^{-1} yr^{-1} per unit disturbance area recorded at F1 for the 2009 water year compared with the 714 kg ha^{-1} yr^{-1} recorded at F2 indicates that natural gas wells can be constructed without significant water quality degradation when necessary erosion control measures are implemented. However, once stream channels are filled in and obliterated, remediative BMPs like silt fence and revegetation are unlikely to have a significant effect in reducing erosion and minimizing aquatic habitat degradation. The stormwater generated by even relatively small rain events washed pollutants directly off the pad into the stream, with no opportunity for deposition and filtration.

Since construction of gas well pads in the state of Texas is not currently regulated like other construction sites, the responsibility for ending the practice of stream channel obliteration for gas well pad construction falls on the industry to self-regulate this practice. There is a precedent for effective industrial self-regulation in Texas, where forest practices like clearcutting along intermittent and perennial streams are not regulated by state or federal environmental agencies. After research demonstrated that clearcutting could have significant impacts on water resources [23], voluntary BMPs that restrict forest harvesting along streams were adopted by the forest industry in Texas by the mid-1980s. After an extensive education and outreach campaign, 98% of forestry activities in Texas voluntarily retained streamside buffers by 2011 [8]. Like the production of wood and fiber, development of natural gas resources is necessary for society. However, this must be conducted with effective and systematic implementation of soil and water conservation practices that ensure these land use changes will not impair surface waters in regions where extensive natural gas development will occur.

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ATTACHMENT C

STUDY 40

frontiers in MICROBIOLOGY

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Assessing impacts of unconventional natural gas extraction on microbial communities in headwater stream ecosystems in Northwestern Pennsylvania

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Hydraulic fracturing and horizontal drilling have increased dramatically in Pennsylvania Marcellus shale formations, however the potential for major environmental impacts are still incompletely understood. High-throughput sequencing of the 16S rRNA gene was performed to characterize the microbial community structure of water, sediment, bryophyte, and biofilm samples from 26 headwater stream sites in northwestern Pennsylvania with different histories of fracking activity within Marcellus shale formations. Further, we describe the relationship between microbial community structure and environmental parameters measured. Approximately 3.2 million 16S rRNA gene sequences were retrieved from a total of 58 samples. Microbial community analyses showed significant reductions in species richness as well as evenness in sites with Marcellus shale activity. Beta diversity analyses revealed distinct microbial community structure between sites with and without Marcellus shale activity. For example, operational taxonomic units (OTUs) within the Acetobacteracea, Methylocystaceae, Acidobacteriaceae, and *Phenylobacterium* were greater than three log-fold more abundant in MSA+ sites as compared to MSA− sites. Further, several of these OTUs were strongly negatively correlated with pH and positively correlated with the number of wellpads in a watershed. It should be noted that many of the OTUs enriched in MSA+ sites are putative acidophilic and/or methanotrophic populations. This study revealed apparent shifts in the autochthonous microbial communities and highlighted potential members that could be responding to changing stream conditions as a result of nascent industrial activity in these aquatic ecosystems.

Keywords: marcellus shale, fracking, 16S rRNA gene sequencing, next generation sequencing, methanotrophs, beta diversity, acidophilic

INTRODUCTION

Headwater streams are central to ecosystem functioning, and they are particularly sensitive to anthropogenic disturbances due to the combination of direct pollutant inputs to the watershed and the transmission of impacts from adjacent riparian terrestrial ecosystems (Sweeney, 1992; Lemke et al., 1997; Pusch et al., 1998; Lemke and Leff, 1999; Maloney and Weller, 2011; Janisch et al., 2012; Webber, 2012; Ding et al., 2013). However, the importance of headwater streams on downstream ecosystem health has only recently received attention. Low-order stream ecosystems provide habitats for unique and local communities, and impact diversity (Meyer et al., 2007) and quality of regional freshwater ecosystems (Peterson et al., 2001; Alexander et al., 2007; Freeman et al., 2007; Wipfli et al., 2007). In particular, aquatic microbial communities are central to energy flow within these ecosystems (Peterson et al., 2001; Findlay et al., 2002; Gulis and Suberkropp, 2003; Hall and Tank, 2003; Puddu et al., 2003; Wright and Covich, 2005; Hall et al., 2012; Schelker et al., 2012). The first biotic response to environmental perturbations can be seen at the lowest trophic levels, as microbial communities can readily respond to changes in their surrounding abiotic environments. Aquatic microbial community structure changes in response to biogeochemical alterations from anthropogenic sources, including agricultural, industrial, and recreational activities (Wassel and Mills, 1983; Clivot et al., 2013; Sun et al., 2013). Despite recent increases in prevalence, the impact of unconventional natural gas extraction, referred to as hydraulic fracturing or fracking, on headwater stream ecosystems has yet to be evaluated.

The mechanics and process of hydraulic fracturing and modern shale gas development has been previously described (Hubbert and Willis, 1954; Arthur et al., 2008; Ground Water Protection Council and ALL Consulting, 2009). Briefly, fracking involves drilling first vertically, then horizontally, toward a gas-bearing formation. Once the horizontal well is drilled, a combination of water, sand, and chemicals is injected at high pressure, fracturing the target formation to efficiently recover natural gas. Recent technological advances and economic conditions have favored the development of gas-bearing shale formations within the United States. When fracking occurs in the Marcellus shale formation, the resulting activity is described as Marcellus shale activity. As one of the largest shale gas formations in the United States, the natural gas extraction from the Marcellus shale has revitalized the energy industry in the northeastern United States and as a result, the state of Pennsylvania has fostered the fastest growing natural gas industry in the United States[1,2] (United States Energy Information Administration, 2012). Accordingly, the rapid development of hydraulic fracturing in Pennsylvania likely will provide a variety of intriguing challenges and opportunities to investigate.

Numerous environmental surveys have postulated that hydraulic fracturing may lead to increased risks for groundwater (Davies, 2011; DiGiulion et al., 2011; Warner et al., 2012; Jackson et al., 2013; Vengosh et al., 2014), surface-water (Øvreås and Jensen, 1998; Entrekin et al., 2011; Barbot et al., 2013; Olmstead et al., 2013), and air pollution (Pacsi et al., 2013; Roy et al., 2014). Surface waters located near shale gas wells have a particularly high risk of being impacted directly or indirectly by natural gas activities (Entrekin et al., 2011; Barbot et al., 2013; Olmstead et al., 2013). Nearly 700 violations were issued by the Pennsylvania Department of Environmental Protection (PADEP) to shale gas companies from 2008 to 2010 for surface water pollution (Entrekin et al., 2011). The effects of spills, well blowouts, and storage leaks on surface water are not well known due to a lack of empirical measurement and due to the variable and unknown composition of fracking fluids (Entrekin et al., 2011; Waxman et al., 2011). Land-use alterations within the Marcellus shale region will also likely have an impact on surface water quality, especially in small headwater ecosystems. Drohan et al. (2012) identifies that core forest, defined as forest situated more than 100 meters from cleared area, is of particular risk to land use changes by Marcellus shale activity through road construction and pad development (Drohan et al., 2012). On average, 1.2–2.0 ha of land are used to construct a wellpad (3.5 ha if additional infrastructure such as roads and pipelines are considered) (Johnson et al., 2010; Grant et al., in press). Changes in stream water quality can occur from increased overland flow associated with forest disturbance (Sollins and McCorison, 1981; Meyer and Tate, 1983; Bryce et al., 2010; Jardine et al., 2012; Schelker et al., 2012; Palviainen et al., 2014). Thus, these alterations in land use within forested headwater ecosystems are likely to have major effects on stream conditions and the communities they support.

[1]Pennsylvania is the Fastest-Growing Natural Gas-Producing State (2013).
[2]Pennsylvania Drives Northeast Natural Gas Production Growth (2011). Available online at: http://www.eia.gov/todayinenergy/detail.cfm?id=2870.

Despite potential environmental disruptions, few investigations have been published that examine the ecology of stream systems in the context of hydraulic fracturing. Further, there are no existing studies investigating the potential effects of fracking on aquatic microbial communities. While some limited metagenomic analyses of fracturing fluids and flowback waters have identified potential microbial contaminants of wells and associated infrastructure (Struchtemeyer et al., 2011; Murali Mohan et al., 2013; Wuchter et al., 2013), nothing is currently known about the impacts of fracking on surrounding environmental microbial communities. In this study, we applied microbial community analysis to headwater stream ecosystems with different histories of fracking, specifically focusing on differences between sites with no fracking activity and those with activity. We used high-throughput sequencing of the 16S rRNA gene to analyze the microbial community structure of water, sediment, bryophyte, and biofilm samples from 26 headwater stream sites in northwestern Pennsylvania. For the first time, this study revealed apparent shifts in aquatic microbial communities impacted by fracking and highlighted potential sentinel taxa that could be responding to changing watershed conditions as a result of Marcellus shale activity.

MATERIALS AND METHODS

SITE SELECTION

Stream study sites were all located on public lands, and appropriate permits were acquired through the Department of Conservation and Natural Resources (http://www.dcnr.state.pa.us) and the Pennsylvania Game Commission, SFRA-1322 (http://www.pgc.state.pa.us/portal/server.pt/community/pgc/9106). Permits were acquired through the PA Fish and Boat Commission (Permit #604) to conduct all aquatic research described. All permits are available upon request at Juniata College.

Twenty-six Pennsylvanian watersheds with unconventional shale gas well permits (PADEP, 2012) were selected for study based on the following criteria. (i) they were forested watersheds with little or no prior anthropogenic activity, (ii) they were headwater streams, (iii) they had naturally-reproducing trout species [*Salvelinas fontinalis* and *Salmo trutta*], and (iv) they were located within the Marcellus shale region in northwestern Pennsylvania. Figure S1 shows stream sampling locations. Streams without fracking infrastructure development prior to sampling were grouped as lacking Marcellus shale activity (MSA−, $n = 10$). Stream sites with at least one wellpad were considered to exhibit Marcellus shale activity (MSA+, $n = 16$), except in the cases of Trout Run and Deer Run. An Unnamed Tributary to Trout Run and Alex Branch are tributaries of Trout Run that exhibit Marcellus shale activity. Because these two MSA+ tributaries feed into Trout Run, it is included in the MSA+ group. Deer Run did not have any drilled wells in its watershed, but wellpad construction began prior to sampling. Therefore, Deer Run was classified as exhibiting Marcellus shale activity. **Table 1** provides information regarding the extent of Marcellus shale activity within the sampled watershed. It should be noted that two streams (Little Laurel Run and Alex Branch) had documented hydrofracking-related contamination occur within their watershed prior to our

Table 1 | Watershed information about streams sampled in this study.

Stream	Impact status*	Number of wells	Number of wellpads	Distance to nearest well (meters)	River basin
Alex branch	MSA+	10	9	1010	West Branch of the Susquehanna
Bear run	MSA+	2	1	1193	Clarion River
Ben's creek	MSA−	0	0	0	Stonycreek River
Big break hollow	MSA−	0	0	0	Juniata River
Camp run	MSA−	0	0	0	Allegheny River
Coldstream run	MSA+	12	5	970	West Branch of the Susquehanna
Crooked run	MSA−	0	0	0	West Branch of the Susquehanna
Dead man's lick	MSA−	0	0	0	West Branch of the Susquehanna
Deer run	MSA+	0	2	0	West Branch of the Susquehanna
Diamond run	MSA−	0	0	0	Juniata River
Dixon run	MSA+	0	2	0	West Branch of the Susquehanna
East beaver run	MSA+	0	0	0	Allegheny River
Findley run	MSA−	0	0	0	Conemaugh River
Indian run	MSA+	12	2	1738	Allegheny River
Iron run	MSA+	2	1	3398	Allegheny River
Laurel run	MSA+	2	1	3188	Clarion River
Lick run	MSA+	10	3	1470	West Branch of the Susquehanna
Little laurel run	MSA+	26	11	1887	West Branch of the Susquehanna
Little wolf run	MSA+	4	2	2481	Clarion River
Long run	MSA+	4	2	2063	Clarion River
South Branch North Fork Redbank Creek	MSA+	1	1	1065	Allegheny River
Stone run	MSA+	19	5	538	West Branch of the Susquehanna
Straight creek	MSA−	0	0	0	Clarion River
Trout run	MSA+	12	2	1430	West Branch of the Susquehanna
Un-named Tributary to the Clarion River	MSA−	0	0	0	Clarion River
Vineyard run	MSA−	0	0	0	Clarion River

**MSA+ denotes presence of Marcellus shale activity, while MSA− represents streams with no Marcellus shale activity in the watershed.*

sampling efforts according to the Pennsylvania Department of Environmental Protection (PADEP, 2012). Detailed information about the sample sites and watershed characteristics are further described in Grant et al. and have been summarized in the Supplementary Information (Grant et al., in press). The similarity of the stream and watershed characteristics of the selected streams make these sites ideal to compare with respect to the impacts of Marcellus shale activity (Grant et al., in press).

FIELD SAMPLING

Samples ($n = 58$) were collected from bryophyte, sediment, biofilm, and water matrices. Aquatic mosses ($n = 24$) were cut directly from submerged rock substrates with a sterile scalpel to collect their microbial communities. Moss samples, consisted of two common water mosses, *Fontinalis sphagnifolia* and *Fontinalis antipyretica*. Sediment samples ($n = 24$) located adjacent to the water-bank interface were collected using sterile scoops. Biofilm samples from South Branch North Fork Redbank Creek and Little Laurel Run ($n = 2$) were collected in sterile 50 mL conical tubes. For water samples ($n = 8$), 1 liter of stream water was collected from a central riffle using a sterile Nalgene bottle. Water samples were filtered on site with 0.22 μm polyethersulfone filters (Millipore, Billerica, MA) and stored in sterile Whirl-Pak bags (Nasco, Fort Atkinson, WI). All samples were immediately placed on dry ice then stored at −80°C. Stream water chemistry measurements [pH, conductivity, salinity, total dissolved solids (TDS), and temperature] were taken on site at the time of sampling with a PCSTestr 35 (Oakton Instruments, Vernon Hills, IL) that was calibrated weekly. For later analysis of organic matter content (DOC) and nitrogen concentration, water samples were collected in pre–cleaned amber glass bottles and pre–cleaned 500 ml polyethylene (HDPE) bottles and stored on ice. Water samples were collected upstream of microbial water samples and at the centroid of flow in riffles under baseflow conditions.

DNA EXTRACTION

Prior to nucleic acid extraction, 0.4 g of bryophyte material from each site was transferred to sterile centrifuge tubes and 4 mL of phosphate buffered saline solution (1X PBS) was added. The samples were vortexed for 15 s and centrifuged at 4000 × g at 4°C for 5 min. The supernatants were centrifuged at 16,000 × g for 10 min at 4°C and the resulting cell pellets were resuspended in 200 μL of 1X PBS. Nucleic acid extractions were performed on bryophyte-derived cell pellets, 0.6 g of sediment and biofilm samples, and water filters using a modified Cetyltrimethyl ammonium bromide (CTAB) Phenol/Chloroform/Isoamyl-alcohol method as described in (Hazen et al., 2010). A more detailed description of the extraction procedure can be found in the Supplementary Information.

ILLUMINA TAG PCR

Duplicate 25 μL Illumina tag Polymerase Chain Reactions (PCR) from each sample ($n = 58$) contained final concentrations of 1X PCR buffer, 0.8 mM dNTP's, 4 μM 515F Illumina barcoded forward primers, 4 μM 806R reverse primers, 0.25 U Taq Polymerase per reaction, and 10 ng of template DNA per reaction. PCR was carried out on a MJ Research PTC-200 thermocycler (Bio-Rad, Hercules, CA) using the following cycling conditions: 98°C for 3 min; 25 cycles of 98°C for 1 min, 55°C for 40 s, and 72°C for 1 min; 72°C for 10 min; and kept at 4°C. PCR products were visualized on a 2% E-gel (Life Technologies, Carlsbad, CA). Library purification, verification, and sequencing of libraries are described in the Supplementary Information.

BIOINFORMATICS AND STATISTICAL ANALYSES

Sequence data for this project can be found in NCBI's Short Read Archive under accession number SRP046758. Due to a quality score drop at 98 bp on reverse reads, only the forward reads were analyzed. Sequences were trimmed at a length of 120 bp and quality filtered at an expected error of less than 1% using USEARCH v7 (Edgar, 2013). After quality filtering, reads were analyzed using the QIIME 1.7.0 software package (Caporaso et al., 2010; Caporaso and Lauber, 2011). Chimeric sequences were identified using USEARCH61 (Edgar, 2010). A total of 3.2 million sequences were retrieved after quality filtering and chimera checking. Open reference operational taxonomic units (OTUs) were picked using the USEARCH61 algorithm (Edgar, 2010), and taxonomy assignment was performed using the Greengenes 16S rRNA gene database (13-5 release, 97%) (DeSantis et al., 2006). Sequences that did not match the database were subsequently clustered using de novo clustering. A detailed description of alpha and beta diversity analyses can be found in the Supplementary Information. Visualization of trends in microbial community structure for MSA+ and MSA− samples were generated in R using the *Phyloseq* (McMurdie and Holmes, 2013; R Core Team, 2014) and details are described in the Supplementary Information.

LEfSe was used to identify taxonomic biomarkers between MSA− and MSA+ communities (Segata et al., 2011) and for intra-matrix comparisons. Genus-level relative abundances were multiplied by 1 million and formatted as described in Segata et al. (2011). Comparisons were made with "Impact Status" (MSA+ or MSA−) as the main categorical variable ("Class") and "Sample Matrix" (sediment, bryophyte-associated, or water) as the secondary categorical variable ("Subclass"). Alpha levels of 0.05 were used for both the Kruskal–Wallis and pairwise Wilcoxon tests. Linear Discriminant Analysis (LDA) scores for the top 10 features from each class were plotted.

Statistical analysis of stream water pH, conductivity, TDS, salinity, and temperature was conducted between MSA+ and MSA− streams using *t*-test and Kruskal–Wallis tests in Minitab (v.16). Data was transformed ($\log_{10}$) to help meet the assumptions of ANOVA, while a non-parametric Kruskal–Wallis test was used for stream pH comparisons. All statistical analyses were considered significant at $\alpha = 0.05$. Pairwise comparisons of bacterial community structure between counties, based on bray-curtis and unweighted UniFrac distance metrics, were generated using Phyloseq on an unrarified OTU table. All samples within the object were merged based on county, after which the "distance" function was used to generate all pairwise comparisons.

Spearman correlations were calculated to examine the relationship between continuous abiotic variables and microbial community composition at the genus-level. Due to inherent correlations among abiotic factors, an appropriate p-value correction was not apparent. Instead, the top 10 most positive and negative correlations between genera and pH and number of wellpads were selected for visualization.

RESULTS

WATERSHED AND STREAM MEASUREMENTS

A comparison of water chemistry revealed that pH was the only significantly different parameter between MSA+ and MSA− streams (median pH: MSA+ = 6.9; MSA− = 7.7; Wilcoxon rank sum test, $p = 0.007$). Two of the 26 sample sites, Little Laurel Run (pH = 4.55) and Alex Branch (pH = 4.88), had documented spills of fracking fluid and had the lowest pH of all watersheds characterized in this study. All other collected measures (conductivity, TDS, salinity, temperature, DOC, total nitrogen, and elevation) did not show a significant difference between MSA+ and MSA− streams (Wilcoxon rank sum test; $p > 0.05$ for all comparisons). Additionally, pH was negatively correlated to the number of wellpads present within a watershed (Spearman's rho = −0.72, $p \leq 0.0001$), but no other measures strongly correlated with number of wellpads (absolute value of Spearmans's rho = 0.43, $p > 0.05$ for all other comparisons). The number of wellpads in MSA+ watersheds ranged from 0–11, and there was a maximum of 26 wells in a given watershed (**Table 1**). Comparison of watershed characteristics showed that watershed land cover (% agriculture, % forested, % wetlands, and forest composition) was not significantly different between MSA+ and MSA− watersheds (Grant et al., in press).

MICROBIAL COMMUNITY RESULTS

Phylum-level community structure for MSA+ and MSA− samples within each sample matrix showed that Proteobacteria was the dominant phylum across all samples (**Figure 1**). No major shifts in phyla abundance were noted between MSA+ and MSA− sites for sediment, water, and bryophyte samples (Figures S2A–S2C). However, major changes in community structure at the phylum rank were observed in the two biofilm samples from Little Laurel Run ($n = 13$ wellpads) and South Branch North Fork Redbank Creek ($n = 1$ wellpad). The biofilm sample from Little Laurel Run, a spill site, was dominated by multiple phyla, including Proteobacteria, Cyanobacteria, Verrucomicrobia, Acidobacteria, and Bacteroidetes, while the biofilm sample from South Branch North Fork Redbank Creek was exclusively dominated by Proteobacteria (Figure S2D).

Alpha diversity rarefaction curves suggested a reasonable coverage of diversity was reached, as curves approached a horizontal asymptote as sequencing depth increased (Figure S3). Based on both observed species and Chao1 alpha diversity metrics, water samples possessed the greatest species richness, followed by sediment, bryophyte, and biofilm samples respectively. Although sequencing depth of water was an order of magnitude lower

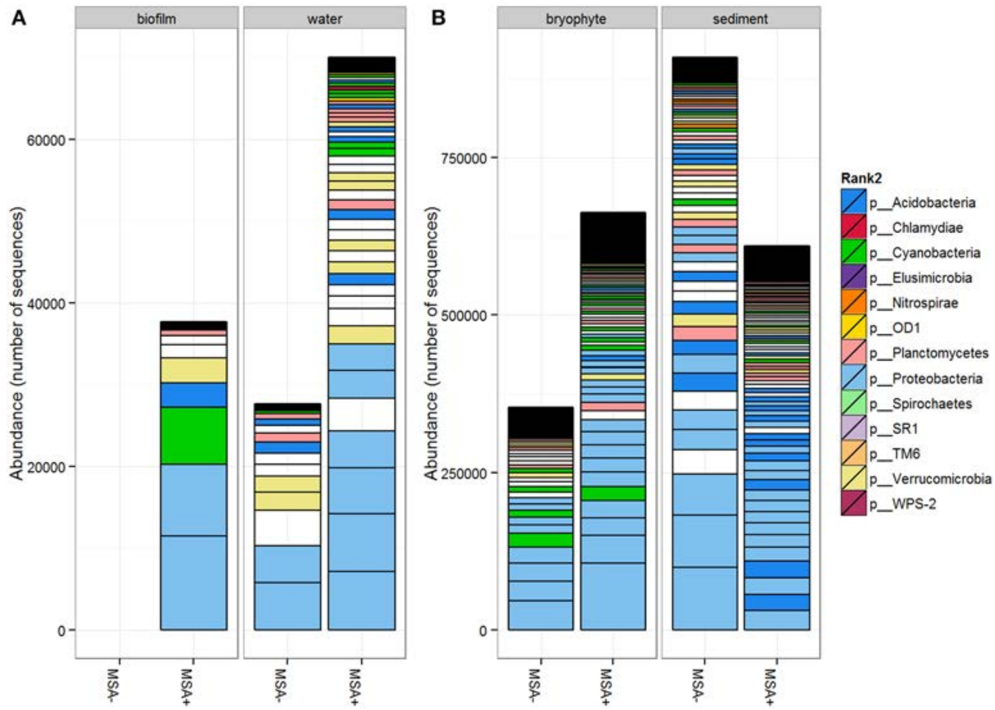

FIGURE 1 | Abundance of bacterial phyla in (A) biofilm ($n = 2$) and water ($n = 8$) samples, (B) bryophyte ($n = 24$), and sediment ($n = 24$) samples. Samples were grouped by Marcellus shale activity status and by sample matrix. OTUs with more than 20 sequences in at least one sample are shown. Plots were created in phyloseq (McMurdie and Holmes, 2013) using the functions tax_glom and tip_glom (height = 0.9) to consolidate the taxa. Unknown phyla are shown in white. All biofilm samples were collected from MSA+ sites. Note that the difference in sequence abundance is attributed to number of samples and sequencing depth.

than for bryophyte and sediment samples, alpha rarefaction plots showed water samples were more diverse at an even sampling depth (Figure S3).

Richness was significantly lower in MSA+ samples as compared to MSA− samples (**Table 2**). The number of observed species for MSA+ sites was 3450 ± 679 OTUs, while MSA− sites had an observed richness of 2858 ± 771 OTUs. Reduced richness in MSA+ samples was statistically significant down to class and phylum taxonomic ranks. Bacterial evenness also appeared to be impacted by Marcellus shale activity, as Heip's evenness measurements were significantly lower in MSA+ samples (**Table 2**). When comparing alpha diversity within sediment, bryophyte, and water matrices, samples from MSA+ had lower alpha diversities than MSA− samples from that same matrix (**Figure 2**). Bryophyte-associated microbial communities from MSA+ samples had significantly lower alpha diversity than MSA− samples (Non-parametric two-sample t-test, $p = 0.021$), while water and sediment alpha diversity comparisons were not statistically different.

Bacterial diversity also shared significant relationships with environmental parameters. For example, the number of wellpads in each watershed had a strong negative correlation to alpha diversity (Spearman's rho = −0.551, $p < 0.00001$) while pH had a strong positive correlation with alpha diversity (Spearman's rho = 0.592, $p < 0.00001$). It should be noted that the number of wellpads present and pH also were negatively correlated to each other (Spearman's rho = −0.72, $p < 0.0001$). Bacterial communities from sites with documented fracking fluid releases (Little Laurel Run and Alex Branch) had much lower alpha diversity as compared to average alpha diversity metrics calculated for MSA+ sites without spills and MSA− watersheds (**Table 3**). For example, the sediment and water samples from MSA− watersheds had nearly two times more observed species as compared to the two spill sites (**Table 3**). As mentioned above, these two spill sites also had the lowest pH of all streams evaluated in this study.

Table 2 | Alpha diversity comparisons of MSA+ and MSA− communities across taxonomic ranks.

Metric	Species	Genus	Family	Order	Class	Phylum
Observed OTUs	0.003*	0.005*	0.001*	0.005*	0.009*	0.027*
Chao1	0.002*	0.003*	0.006*	0.003*	0.027*	0.088*
Heip's evenness	0.015*	0.054	0.12	0.169	0.253	0.743

**Indicates significant p-values non-parametric two sample t-test with 999 Monte Carlo permutations, $\alpha = 0.05$.*

Beta diversity analyses of microbial communities revealed distinct microbial community structure between MSA+ and MSA− samples (**Figure 3**). A directional PCoA plot generated using weighted Unifrac distances showed distinct clustering of samples based on number of wellpads (**Figure 3A**) (Vázquez-Baeza et al., 2013). Samples with a high number of wellpads clustered

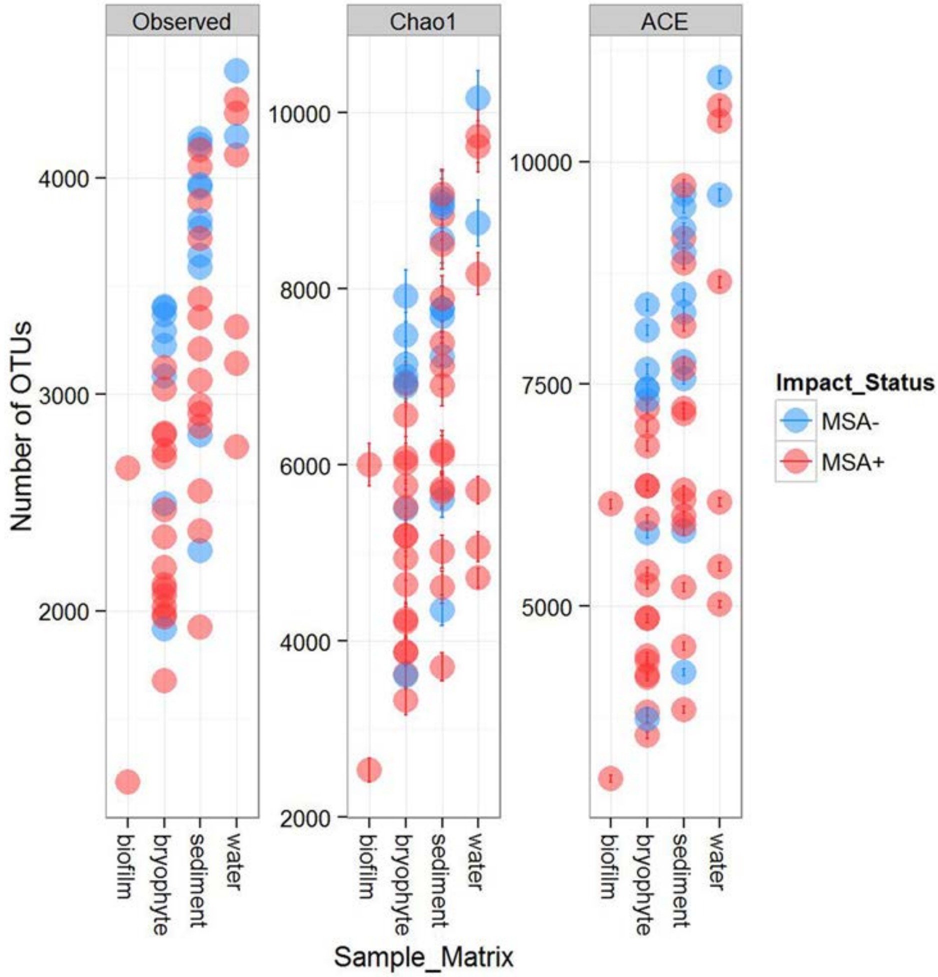

FIGURE 2 | Alpha Diversity metrics in biofilm, bryophyte, sediment, and water samples from MSA+ and MSA− sites. Species richness was estimated by performing multiple rarefactions up to a depth of 11,200 sequences per sample. The richness of a single OTU table from the maximum rarefaction depth was estimated using observed richness, Chao1, and ACE and visualized using Phyloseq. For each metric, species richness is separated by impact status and sample matrix. Samples from impacted environments (labeled MSA+) tend to be less diverse than those from unimpacted ones.

Table 3 | Alpha diversity measures for biofilm sediment and water samples collected from streams with fracking spills.

Sample matrix	Watershed	Group	Heip's evenness	Observed Species	Chao1
Biofilm	Little Laurel Run	Spill	0.0691	2587	5122.05
	SBNFRC	MSA+	0.0079	1210	2657.26
Sediment	Alex Branch	Spill	0.1092	2378	4645.48
	Average (±SD)	MSA+	0.1437 ± 0.0172	3157.67 ± 342.11	6520.81 ± 847.95
	Average (±SD)	MSA−	0.1622 ± 0.0172	3660.91 ± 338.87	7659.47 ± 899.97
Water	Little Laurel Run	Spill	0.1204	2762	5045.77
	Average (±SD)	MSA+	0.1629 ± 0.0246	3746 ± 557.16	7112.29 ± 2081.44
	Average (±SD)	MSA−	0.1698 ± 0.0414	4377 ± 214.14	9578.97 ± 759.96

together. Samples with less wellpads cluster higher on the principal coordinates axis 1, a region where no samples with a high wellpad count are observed. A directional PCoA plot with a strong pH gradient displayed distinct clustering of MSA+ and MSA− samples, suggesting that bacterial community structure is shaped by both pH and fracking status of that watershed (**Figure 3B**). Because sample matrix explained the most (38%) variation in beta diversity across all samples (adonis; Pr $>$ $F = 0.001$),

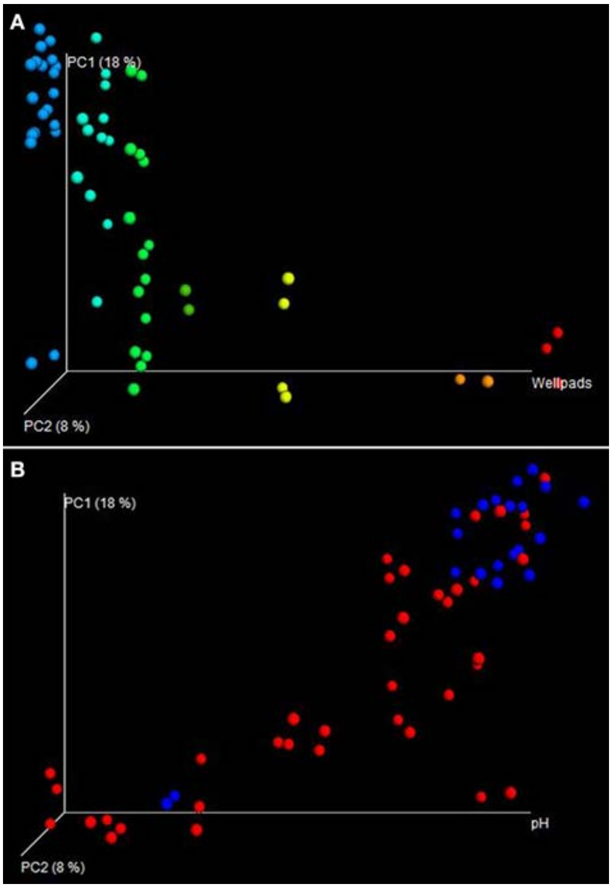

FIGURE 3 | Directional Principal Coordinates Analysis (PCoA) plots were used to visualize differences in weighted UniFrac distances of MSA+ and MSA− samples. (A) Samples were plotted according to number of wellpads along the horizontal axis of the directional PCoA plot. Samples with no wellpads are colored in blue, whereas samples with the highest wellpad count are colored red. Distinct clustering can be observed between samples with a high number of wellpads and samples with a low number of wellpads. A majority of samples with a low wellpad count cluster at the top of the PC1 axis, a region where no samples with a high wellpad count are observed. **(B)** When imposing pH to the horizontal axis, distinct clustering between MSA+ (red) and MSA− (blue) is observed, implying pH in conjunction with impact status shapes microbial community structure.

within-matrix beta diversity statistics were performed. When analyzing each sample matrix separately, sediment (ANOSIM; $p = 0.038$) and bryophyte (ANOSIM; $p = 0.016$) matrices showed distinct clustering of MSA+ and MSA− samples (Figure S4). The number of wellpads accounted for a significant portion of variation in all sample matrices. Within sediment, bryophyte, and water samples the number of wellpads in a watershed explained 20.5% (Adonis; $\Pr > F = 0.002$), 14.2% (Adonis; $\Pr > F = 0.001$), and 20.3% (Adonis; $\Pr > F = 0.024$) of variation in beta diversity, respectively. pH also explained a large amount of variation in beta diversity, as it accounted for 25.8% (Adonis; $\Pr > F = 0.001$), 20.4% (Adonis; $\Pr > F = 0.001$), and 28.8% (Adonis; $\Pr > F = 0.001$) in sediment, bryophyte, and water matrices, respectively.

While differences in microbial community structure were not seen between MSA+ and MSA− sites at the phylum rank, significant changes were observed at a finer phylogenic resolution. For example, biomarker analyses revealed that Methylocystacea, Acetobacteraceae, *Phenylobacterium*, Acidobacteriaceae and WPS-2 groups were amongst the most significantly enriched taxa in MSA+ samples (**Figure 4**). Biomarker analysis performed on individual matrices also supported similar trends (Figures S5–S7). Methylocystaceae, Acetobacteraceae, *Phenylobacterium* were > 3 $\log_{10}$-fold more abundant, and WPS-2 and Acidobacteriaceae were > 2 $\log_{10}$-fold more abundant in MSA+ streams. Five unknown taxa within the order Myxococcales were >3 $\log_{10}$-fold higher in abundance in MSA− streams.

Many of these enriched bacterial taxa also shared strong relationships with both pH and the number of wellpads in a watershed. Spearman correlations revealed a negative relationship between the Acidobacteriaceae and pH and a positive relationship of this taxa with the number of wellpads (**Figure 5**). In all cases, OTUs strongly correlating with wellpads correlated negatively to pH and total nitrogen concentrations (pH absolute Spearman's rho = 0.75–0.58; wellpad absolute Spearman's rho = 0.44–0.57), including unclassified Methylocystaceae, Armatimonadia, Sphingobacteriaceae, Candidatus *Solibacter*, Chthonomonadaceae, Solibacteraceae, Caulobacteraceae, Acidobacteriaceae, Isosphaeraceae, Soilbacteraceae, two genera within the Acetobacteraceae, *Phenylobacterium*, and *Telmatospirillum* (Table S1). Furthermore, OTUs correlating negatively with wellpads correlated positively to pH and total nitrogen concentration (pH absolute Spearman's rho = 0.63–0.73; wellpad absolute Spearman's rho = 0.66–0.59) including unclassified Saprospiraceae, Alteromonadales, Pedosphaerales, Betaproteobacteria, Myxococcales, Anaerolineae, Xanthomonadaceae, Bacteroidetes, and the genera *Devosia*, *Rhodobacter*, *Niabella*, and *Flavobacterium*.

Comparing the impact of spatial variation on the microbial community structure showed that there was not much clustering of the samples by county with the exception of Clearfield and Elk counties (Figure S8). Clearfield county, which contains both spill sites, and nine MSA+ sites, also feeds into streams in Elk county (Figure S1). Both of these counties possessed similar microbial community structures, and were different compared to all other counties based on both bray-curtis and unweighted unifrac metrics (Table S2). In addition, samples collected from both Elk and Clearfield counties were most dissimilar from all other counties sampled in this study.

DISCUSSION

Twenty-six headwater stream ecosystems in Pennsylvania were studied to evaluate the potential impacts of unconventional natural gas extraction on aquatic microbial communities. High-throughput sequencing data enabled deep coverage of the diverse microbial communities in these aquatic environments, which facilitated a detailed analysis of the variation in bacterial diversity and community structure along environmental gradients. As detailed below, microbial community analyses revealed marked differences in bacterial diversity in watersheds with and without fracking. Additionally, the relative abundance of certain bacterial

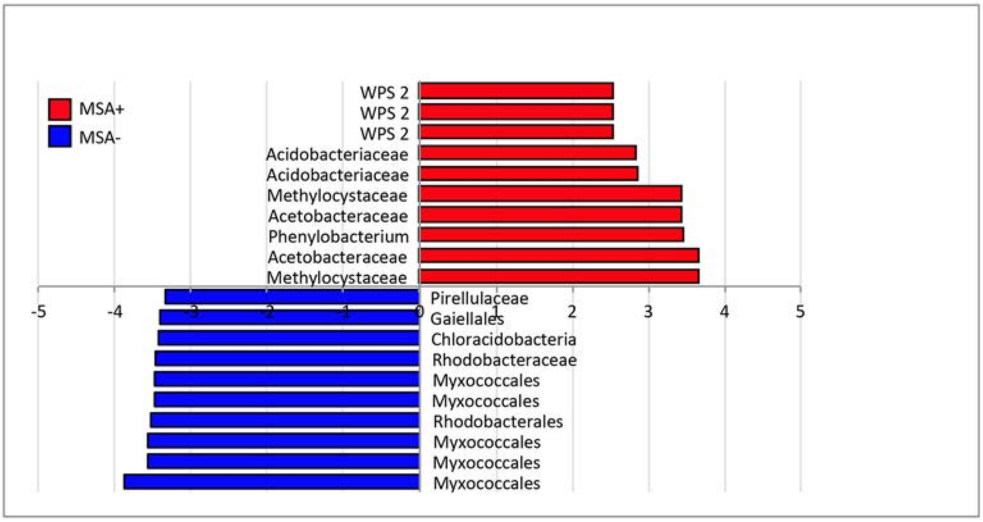

FIGURE 4 | LEfSe plot of taxonomic "biomarkers" of MSA+ and MSA− communities. Here, the top 10 most differentially significant taxa of each group (MSA+/MSA−) are plotted, where red bars represent taxa significantly enriched in MSA+ sites and blue bars signify taxa more abundant in MSA− streams. Features plotted on a logarithmic scale according to the experimental group to which they were significantly associated. LEfSe utilizes Kruskal–Wallis tests to determine significantly different taxonomic features ($\alpha \leq 0.05$) between experimental groups, a pairwise Wilcoxon rank sum statistic to test biological consistency across subgroups ($\alpha \leq 0.05$), and finally a linear discriminant analysis to determine the effect size, or magnitude of variation of the features between groups. Features are plotted on a logarithmic scale according to the experimental group to which they are significantly associated.

taxa correlated with pH gradients and the amount of fracking development in a given watershed.

The stream water chemistry data revealed that streams located in watersheds with Marcellus shale activity had significantly lower pH than sites with no activity. Observed differences in stream pH can be attributed to variation in watershed characteristics, disparities in acid rain deposition, or fracking activities (Grant et al., in press). A thorough GIS survey revealed no significant difference in watershed characteristics in the sites evaluated in this study (Grant et al., in press). Increased acidification by atmospheric deposition would have resulted in concomitant increases in stream water nitrogen concentration, which was not significantly different between MSA+ and MSA− watersheds (Baker et al., 1996; Grant et al., in press). The lower observed pH in watersheds with Marcellus shale activity might be attributed to exposure of pyritic geological formations by the drilling process, as weathering of acid rock has attributed to stream acidification in other scenarios (Hammarstrom et al., 2005; Fierer and Jackson, 2006). Further, a number of concentrated acids (and bases) can be used in the hydraulic fracturing process, and fracking fluid mixtures can themselves be highly acidic, though many of these fluid formulations still remain unknown or undisclosed by industry. The negative correlation between number of wellpads within a watershed and pH suggests that fracking may be directly or indirectly increasing the acidity of headwater stream ecosystems. Further, the fact that no other measured stream characteristics, other than pH, differed between MSA+ and MSA− sites, suggests that our analyses are accurate reflections of the effects of activity, not artifacts related to differences in the sites selected for this study.

Alpha diversity analyses indicated lower diversity in aquatic bacterial communities in streams with Marcellus shale activity as compared to streams with no activity (**Figure 2** and **Table 2**). When comparing alpha diversity within each sample matrix, bryophyte samples had the most significant difference between MSA+ and MSA− sites. Reduction in bacterial richness associated with these bryophyte samples could be related the sensitivity of moss to environmental perturbations, as several moss species are common bio-indicators of environmental quality. It is possible that fracking is indirectly affecting alpha diversity by directly affecting pH. Although not well studied in headwater streams, this finding is supported by previous research, which showed acidic pH was associated with lower bacterial diversity in soils (Baker et al., 1996; Hammarstrom et al., 2005). While this is the first study to assess the impacts of fracking on bacterial communities in aquatic environments, numerous other studies have demonstrated the detrimental impacts a variety of anthropogenic activities have on bacterial diversity in the environment (Wassel and Mills, 1983; Clivot et al., 2013; Sun et al., 2013). However, it should be noted that other environmental factors not measured in this study could also be contributing to decreased observed alpha diversities.

Significant differences in beta diversity were also observed between MSA+ and MSA− sites, suggesting fracking maybe impacting microbial community structure in these aquatic environments. When comparing beta diversity within each sample matrix, bryophyte samples observed the greatest difference in phylogenetic distance between MSA+ and MSA− samples. (Figure S4). This, in congruence with the distinct differences in alpha diversity for bryophyte samples, further suggests that the

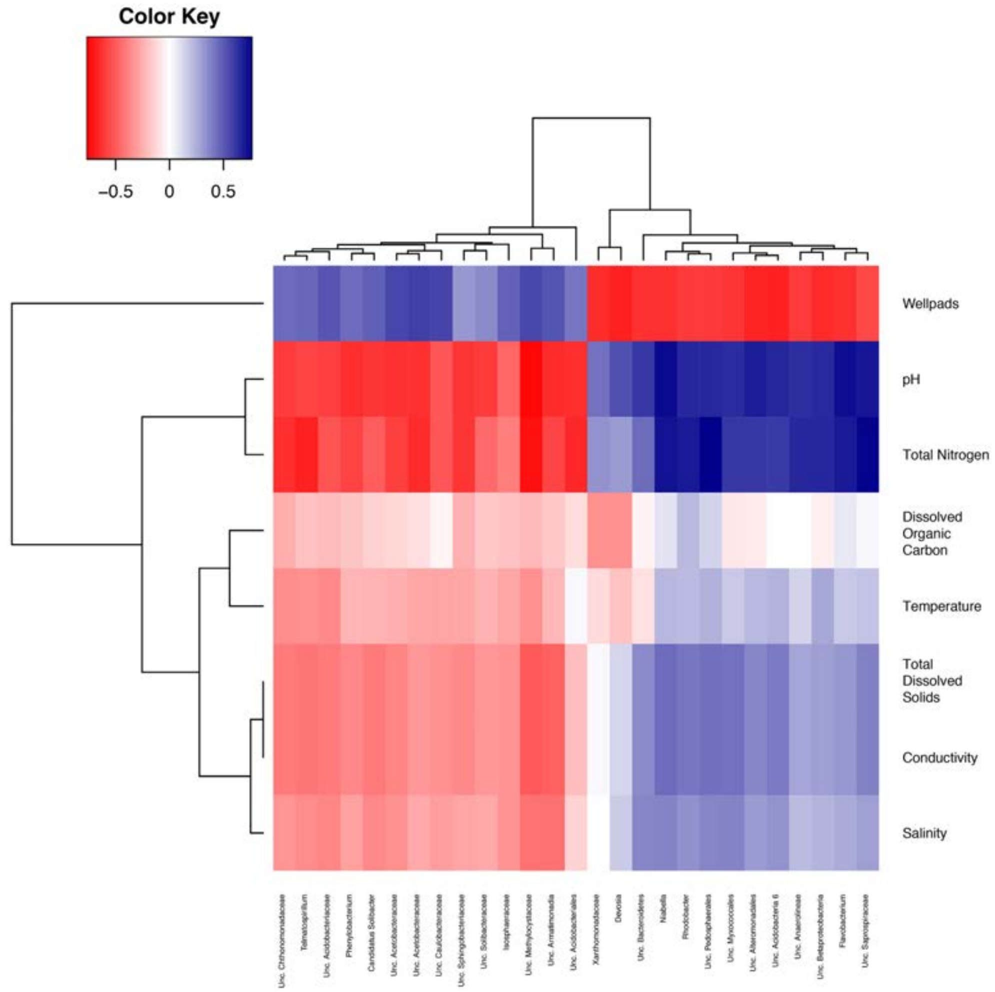

FIGURE 5 | Spearman correlation of abiotic factors with bacterial genera. Spearman correlations were calculated between each genus and several abiotic factors. This heatmap displays the top 10 genera most positively and most negatively correlated with each pH and number of wellpads, respectively. Each cell represents the Spearman rho value for the correlation between a genus (columns) and abiotic factor (rows) ranging from red (for negative correlations) to blue (for positive correlations) with small absolute rho values represented by white. Hierarchical clustering was used to place each genera and abiotic factors with similar relationships near each other.

microbial community associated with bryophyte may be sensitive to potential perturbations in these environments. As aquatic microbial communities are central to ecosystem functioning in headwater streams (Peterson et al., 2001; Findlay et al., 2002; Gulis and Suberkropp, 2003; Hall and Tank, 2003; Wright and Covich, 2005; Hall et al., 2012; Schelker et al., 2012), it is imperative to track the potential response of these aquatic microbial communities to Marcellus shale activities. The number of wellpads in a watershed shared a strong relationship with beta diversity, suggesting increasing development could be shaping microbial community structure in headwater streams via increased land alteration or potential fracking fluid releases. Detailed investigation revealed specific taxa, including the Methylocystaceae, Acetobacteraceae, WPS-2, and *Phenylobacterium* were enriched in MSA+ sites (**Figure 4**). Interestingly, several of these taxa also shared strong correlations with pH and number of wellpads in a watershed, suggesting the relationship of these taxa to environmental changes (**Figure 5**). Increased acidity is known to impact aquatic ecosystem structure at microbial (Mulholland et al., 1992; Dangles and Gessner, 2004; Rousk et al., 2010) and higher trophic levels (Mulholland et al., 1992; Fierer et al., 2007; Simon et al., 2009; Rousk et al., 2010) and may be the mechanism responsible for the increased abundance of potential acidophilic taxa identified in MSA+ sites, such as the Acidobacteriaceae, Acetobacteraceae, and Methylocystaceae. This is particularly clear when considering that the two of the most acidic streams (contaminated by spills) have very high abundance of acidobacterial OTUs, including the Koribacteraceae (11%) and acid-tolerant iron-oxidizers of the genus *Gallionella* (6%).

Interestingly, many of the taxa enriched in MSA+ sites (i.e., OTUs within the Methylocystaceae and Acetobacteraceae) also have methanotrophic capabilities (McKnight and Feder, 1984; Liebner et al., 2009). Recent studies have reported increased groundwater methane concentrations with proximity to natural gas drilling and hydraulic fracturing sites (Davies, 2011; Jackson et al., 2013). Streams in these watersheds are primarily first order streams and are thus fed mostly by groundwater. Increases in abundance of methanotrophic bacteria may be an early indication of increases in methane contamination. It should also be noted that there were increases in the abundance of WPS-2 taxa in MSA+ samples, and this group has been shown to co-reside with methanotrophs, suggesting WPS-2 populations might be utilizing the derivatives of methane oxidation (Uhlig et al., 1986; Nogales and Moore, 2001; Fuss and Smock, 2003; Sharp et al., 2012; Grasby et al., 2013). While high-throughput 16S rRNA gene sequencing provided insight into the potential impacts of fracking on microbial communities in the headwater stream ecosystems, future studies should address the functional capacity and metabolic response of these microbial communities to environmental perturbations associated with fracking. Future studies should focus on measuring methane concentration and using stable isotope probing to determine if the source of this methane is from natural gas stores. In addition, functional meta-omics studies will help describe the functional response of microbial communities to methane in these environments. Metagenomic, metatranscriptome, and metabolomic approaches could provide high-resolution information about the functional capacity, expression and metabolic capabilities of populations enriched and inhibited by fracking activity.

Phenylobacterium was found to be significantly enriched in MSA+ sites and also was positively correlated with the number of wellpads and negatively correlated with pH. Optimal growth of some species of *Phenylobacterium* has occurred on artificial compounds such as chloridazon, antipyrin, and pyramidon, though it is a common inhabitant of soil communities (Eberspacher and Lingens, 2006; Oh and Roh, 2012). Several members of this genus also grow optimally in slightly acidic conditions between pH 6 and 6.5 (Yang et al., 2014). *Phenylobacterium* may be capable of degrading phenyl-compounds and other complex hydrocarbons in acidic environments and are succeeded by other groups of hydrocarbon degraders (Oh and Roh, 2012; Marušincová et al., 2013). Interestingly, this genus was strongly positively correlated with the number of wellpads in a watershed and negatively correlated to pH. Altogether, shifts in several aforementioned taxa suggest that these populations may be responding to environmental perturbations introduced by fracking development either through land disturbances introduced by infrastructure development or potential releases of fracking fluids into the environment. However, future time-series studies will need to carefully track the potential successional changes in microbial communities in response to documented spills.

Several bacterial taxa had significantly lower relative abundance in MSA+ samples, including several OTUs within the Rhodobacteraceae, Myxococcaceae, Hyphomicrobaceae, and Xanthomonadaceae. These taxa shared very strong positive correlations with pH and strong negative correlations with the number of wellpads in a watershed, indicating these taxa may be inhibited by perturbations introduced by fracking development in these watersheds. Interestingly, these taxa are known to have denitrifying capabilities, and lower pH can result in reduction of denitrification rates (Baeseman et al., 2006). Thus, the more acidic pH associated with MSA+ sites could be one possible mechanism for the lower abundance of denitrifying populations observed in MSA− sites.

Beta diversity analyses revealed clear clustering of samples by matrix, which indicates the shifts in community structure are environment-specific. This finding is not surprising, as each microenvironment likely harbors unique conditions in which different populations thrive. When evaluating beta diversity within each matrix, samples were differentiated by Marcellus shale activity, further suggesting fracking activities could be shaping the structure within different aquatic microbial communities. Members exclusively from the Acidobacteria were enriched in sediment samples collected from MSA+ sites, while Alphaproteobacteria were enriched in bryophyte- and water-associated MSA+ samples (Figures S5–S7). As previously mentioned, several enriched taxa within these classes, share negative correlations with pH and have been previously shown to be acidophilic or acid-adapted taxa. Four members of the phylum OD1 were significantly more abundant in MSA+ water samples and the phylum OD1 has been associated with complex hydrocarbon degradation and biofilm formation (Kantor et al., 2013).

While several interesting differences were noted in diversity and bacterial community structure in sites with and without fracking activity, future studies need to address both temporal and spatial variation. An analysis of spatial variation impacts on microbial community structure showed little county-specific clustering of samples, except for streams in the two counties (Clearfield and Elk County), which encompassed streams impacted directly by, or connected to streams that experienced documented fracking fluid spills (Figure S8). It should also be noted that Clearfield County, which contained the highest number of MSA+ sites ($n = 9$), was most dissimilar to all other counties sampled in this study, further illustrating the potential impacts of fracking on microbial community structure (Table S2). While no significant differences were noted in watershed characteristics between MSA+ and MSA− sites, future studies will assess pre- and post-fracking impacts within the same stream to control for any unmeasured differences in watershed characteristics.

CONCLUSIONS

This study represents the first investigation of the potential impact of Marcellus shale activity on aquatic bacterial communities in headwater stream ecosystems in northwestern Pennsylvania. The results of this study showed (i) reduction in the diversity of bacterial communities in streams with fracking activity and (ii) specific shifts in bacterial community structure were indicative of watershed status and correlated with changes in pH. These findings are relevant and timely, as pristine headwater stream ecosystems may bear the largest likelihood for environmental impacts and ecosystem disruption, chiefly because of the potential for relatively large land use alterations introduced in these watersheds as a result of wellpad development. While additional investigation and long-term studies will be necessary to fully elucidate the impacts of Marcellus shale activity on aquatic ecosystems, this study serves to provide baseline bacterial community data for future studies. Future work should focus on additional chemical measurements including isotopic carbon measures, and meta-omics analyses, which will help describe the functional response of microbial communities to potential environmental perturbations introduced by Marcellus shale activities. This study highlighted the potential impacts that fracking can have on headwater stream microbial communities and suggests that additional environmental studies are warranted to more fully characterize and integrate the potential environmental impacts at all trophic levels in these ecosystems.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: http://www.frontiersin.org/journal/10.3389/fmicb.2014.00522/abstract

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Response of Aquatic Bacterial Communities to Hydraulic Fracturing in Northwestern Pennsylvania: A Five-Year Study

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Horizontal drilling and hydraulic fracturing extraction procedures have become increasingly present in Pennsylvania where the Marcellus Shale play is largely located. The potential for long-term environmental impacts to nearby headwater stream ecosystems and aquatic bacterial assemblages is still incompletely understood. Here, we perform high-throughput sequencing of the 16 S rRNA gene to characterize the bacterial community structure of water, sediment, and other environmental samples (n = 189) from 31 headwater stream sites exhibiting different histories of fracking activity in northwestern Pennsylvania over five years (2012–2016). Stream pH was identified as a main driver of bacterial changes within the streams and fracking activity acted as an environmental selector for certain members at lower taxonomic levels within stream sediment. Methanotrophic and methanogenic bacteria (i.e. Methylocystaceae, Beijerinckiaceae, and *Methanobacterium*) were significantly enriched in sites exhibiting Marcellus shale activity (MSA+) compared to MSA− streams. This study highlighted potential sentinel taxa associated with nascent Marcellus shale activity and some of these taxa remained as stable biomarkers across this five-year study. Identifying the presence and functionality of specific microbial consortia within fracking-impacted streams will provide a clearer understanding of the natural microbial community's response to fracking and inform *in situ* remediation strategies.

Increasing global reliance on natural gas is a critical issue that has many economic and environmental implications. On average, the global utilization of natural gas exceeds 120 trillion cubic feet (Tcf) per year and is expected to increase at an astounding rate to total 203 Tcf by 2040[1]. In the past decade, technological development has informed many methods of natural gas extraction, as it has become the primary fuel source for energy generation and residential/commercial heating[1,2]. Combinations of horizontal drilling and hydraulic fracturing (fracking) processes have revolutionized the industry by opening up new areas for oil and gas development that were previously inaccessible within the U.S.[3] The Devonian age (416−359.2 My) Marcellus Shale formation is the largest shale reserve[4], producing 40% of the U.S. shale gas[5] and is positioned within the Appalachian Basin, located approximately 1,219–3,000 m below the surface[6]. Production data estimate that as much as 489 Tcf of recoverable resources are contained within the expanse of the Marcellus Shale formation[4,7].

Briefly, fracking methods involve first drilling vertically, then horizontally, toward the subterraneous gas-bearing formation. Large volumes of fracking fluid, typically composed of water (90%) mixed with sand (9%) and chemical additives (1%), are injected into each well at high pressures to open and enlarge the fractures within the shale formation[8–12]. After the initial fracture process, internal pressure of the rock formation causes fracking fluid in addition to brines, metals, organic compounds, and radionuclides to return to the surface as flowback fluid[3,13,14]. As the well matures, produced water characterized by hydrocarbons, remaining fracking fluid, subsurface brines, and formation solutes is brought to the surface for most of the well life[11,14].

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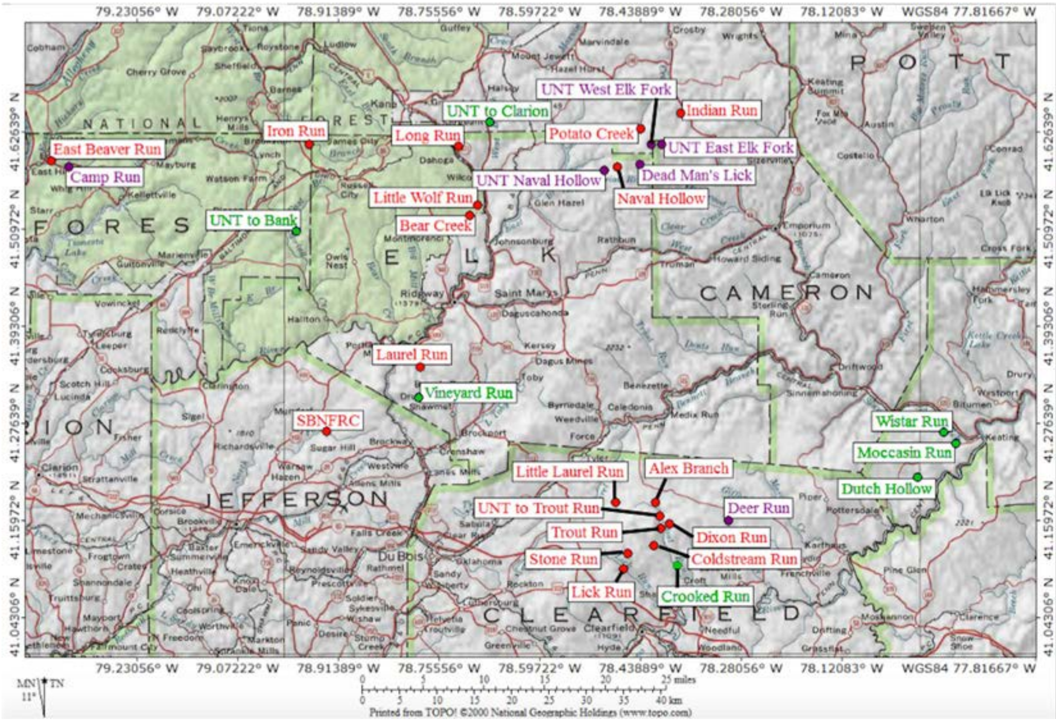

Figure 1. Map of Stream Sites. Stream sites (n = 31) across Clearfield, Jefferson, Forest, Elk, Cameron, and Mckean counties of northwestern Pennsylvania. TOPO! Version 2.6.4 of National Geographic Holdings (https://shop.nationalgeographic.com/category/topo-state-series) was used to generate the map. Red represents MSA+ streams, green represents MSA− and purple represents streams that changed from MSA− to MSA+ during the sampling period (2012–2016). Two streams Findley Run and Diamond Run within southern Blair and Cambria counties not shown on this map are also classified as MSA− streams.

Research within the last few years has postulated that fracking activity poses critical risks to public health[15–17] and the environment[18–20]. Fracking practices including treatment of flowback can lead to increased environmental risks for groundwater[21,22], surface water[20,23,24], and air pollution[25,26]. In fact, from 2009 to 2015 there were a reported 490 violations connected to improper handling of fracking residual waste and 292 violations for failing to adopt pollution prevention when handling fracking wastewater[27]. Because unconventional methods of natural gas extraction are connected to a reported 30% increase in methane emissions compared to conventional wells, the potential migration of methane into groundwater and the atmosphere is a prominent concern[28]. Alterations in land-use associated with fracking development, mismanagement of fracking fluids, and potential environmental contamination of flowback constituents present possible risks to forested headwater stream ecosystems[29,30]. Small headwater streams are particularly vulnerable to direct pollutant inputs as well as disturbances within nearby riparian terrestrial environments[29].

While some studies have begun to assess environmental impacts of fracking[31–36], recent studies have also indicated that robust microbial communities exist within fracking-associated fluids[37,38]. Microbial communities are integral to degrading and metabolizing many of the complex compounds found in the injected and flowback fluids. For example, halotolerant bacteria associated with hydrocarbon oxidation, fermentation, and sulfur-cycling metabolisms including the genera *Halanaerobium*, *Halomonas*, *Vibrio*, *Halolactibacillus*, *Marinobacter*, and autotrophs belonging to *Arcobacter* comprise >90% of the microbial communities within flowback and produced fluids[39]. Moreover, recent studies have indicated that streams within proximity to fracking activities have undergone shifts in their bacterial community structure[35]. For example, Methylocystacea, Acetobacteraceae, *Phenylobacterium*, and Acidobacteriaceae and an increase in methanotrophic bacteria abundance were linked to Marcellus shale activity[40].

Additional investigations are necessary to monitor aquatic microbial communities and their associated functionality regarding long-term fracking operations. Long-term temporal changes to bacterial structure, water quality, and stream characteristics have yet to be evaluated, especially individual streams that have transformed from pre-fracked to post-fracked. Notably, since 2011, Pennsylvania is second only to Texas in the amount of producing gas wells[41]. This study investigated the bacterial community profiles of 31 headwater stream ecosystems in northwestern PA (Fig. 1) exhibiting different histories of fracking over the course of five years (2012–2016). Streams categorized as having Marcellus Shale activity (MSA+) included streams nearby infrastructure for fracking operations (i.e. wellpads) or active fracking. MSA− streams were not proximal to any associated fracking operations through the duration of the study. For the first time, a temporal investigation of fracking impact on microbial communities within the same watershed was possible with MSA− to MSA+ status changes of six

MSA+ Streams (n = 16)	MSA− Streams (n = 9)	MSA− to MSA+ Streams (n = 6)[a]
Alex Branch	Crooked Run	Camp Run (2014)
Bear Creek	Diamond Run	Dead Man's Lick (2014)
Coldstream Run	Dutch Hollow	Deer Run (2013)
East Beaver Run	Findley Run	UNT to Naval Hollow (2015)
Indian Run	Moccasin Run	UNT East Elk Fork (2015)
Iron Run	UNT to Bank	UNT West Elk Fork (2015)
Laurel Run	UNT to Clarion	
Lick Run	Vineyard Run	
Little Laurel Run	Wistar Run	
Little Wolf Run		
Long Run		
Naval Hollow		
Potato Creek		
Stone Run		
SBNFRC		
Trout Run		

Table 1. Stream Classification by Impact Status. [a]Year of status change is included in parentheses.

streams. Together, these data permit a powerful and robust identification of biomarker taxa for fracking activity to quantify the broader environmental consequences of fracking operations.

Results

Watershed and Stream Measurements. Results indicate that stream pH was significantly different between MSA+ and MSA− streams over all 5 years (median pH: MSA+ = 6.54; MSA− = 7.15; Wilcoxon rank sum test, $p = 0.001$). Two stream sites, Alex Branch and Little Laurel, which had documented spills prior to sampling, had the lowest median pH levels, 4.96 and 4.5, respectively. Table S1 displays pH, number of active wells, wellpad count, and impact status of each site by year. Stream pH was negatively correlated with number of active wells within the watershed (Spearman's rho = −0.53, $p < 0.001$) but no other measures (TDS, salinity, dissolved oxygen, temperature, collection year, and wellpad count) shared a strong relationship with stream pH (Table S3). The number of nearby active wells did not significantly correlate with any other measured parameters (Absolute Value Spearman's rho = 0.23, $p < 0.001$). Our previous study identified that watershed land cover (% agriculture, % forested, % wetlands, and forest composition) was not significantly different between these MSA+ and MSA− watersheds in 2013[36]. It is important to note that six streams changed from MSA− to MSA+ during the sampling period and by 2016, only nine streams were classified as MSA− streams (Table 1).

Environmental Drivers of Bacterial Community Composition. Partial-least squares linear discriminant analysis (PLS-DA) model revealed that certain environmental parameters could be contributing to microbial community variation within sediment samples (n = 86). Sediment samples from sites with the most active wells (21 active wells) were significantly separated from those with zero active wells, revealing a variation in bacterial communities based on the number of active wells in the watershed (Fig. 2). In the PLS-DA score plot, two axes of variation (t1 and t2) were calculated with the R^2X, R^2Y, and Q^2 parameters of 0.509, 0.935, and 0.0503, respectively (Fig. 2). However, pH (in addition to all other measured water parameters) did not significantly explain differences in the comprehensive microbial composition (sediment, water, bryophyte, biofilm) (Adonis, $R^2 = 0.01$, $p < 0.022$) or within just sediment samples (Adonis, $R^2 = 0.11$, $p < 0.0017$). Stream sediments were not found to be statistically different between MSA+ and MSA− status in each year (Table S4).

Bacterial Community Structure and Diversity. Phylum-level community structure for MSA+ and MSA− samples within each sample matrix revealed Proteobacteria as the dominant phylum (32–71%) across all samples over time. Sediment samples were largely composed of Acidobacteria (6–51%) and biofilm samples ranged in community composition, but Proteobacteria remained the dominant phylum (35–50%) (Fig. S1). Biofilm samples from Little Laurel (LiLR) and South Branch North Fork Redbank Creek (SRC) had high abundances of sequences matching Cyanobacteria (30–40%) (Fig. S2). No major shifts in microbial composition at the phylum level were observed in both MSA+ and MSA− across all sampling years.

Alpha rarefaction curves suggested a reasonable coverage of diversity was reached (Fig. S3). Bryophyte samples were collected only in 2012 and 2013, so they were omitted from downstream temporal analyses. Sediment samples possessed the greatest richness followed by water and biofilm, respectively (Fig. S4). Across all samples, there were no significant correlations between alpha diversity metrics and MSA status (Table S5). Further, for each year, sediments from MSA+ and MSA− sites did not have significantly different richness (Non-parametric two-sample *t*-test, $p > 0.05$).

Beta diversity of weighted Unifrac distances revealed the possible shaping of bacterial assemblages within members of lower taxonomic ranks. Sample matrix explained the most variation (33.38%) in beta diversity across all samples (Adonis, $p < 0.001$) (Fig. S5). Further analyses investigated exclusively sediment samples (n = 86), as

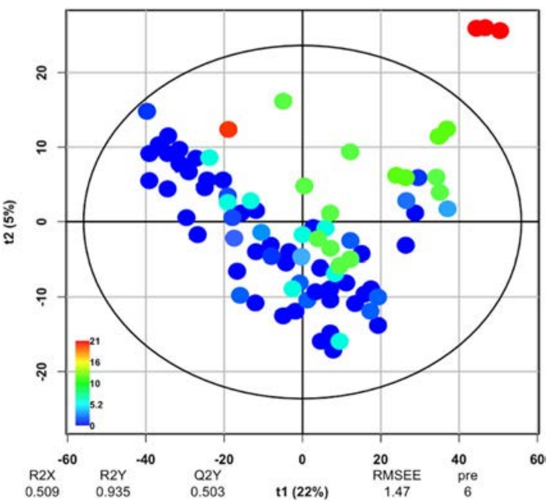

Figure 2. Partial least-squares discriminant analysis (PLS-DA) obtained from sediment samples (n = 86) of both MSA+ and MSA− stream sites. PLS-DA analyses were performed with a CSS normalized OTU table and scores representing each sample are plotted on a PCA plot. Red signifies samples from sites with 21 active wells, and blue signifies zero active wells. The ellipse surrounding the majority of the scores is Hotelling's T^2 elliptical tolerance region, which indicated the 95% confidence limits. There is a clear separation of samples between samples with 10–21 active wells and with 0–5 active wells based on the model quality parameters: $R^2X = 0.509$, $R^2Y = 0.935$, and $Q^2 = 0.503$. These significant values represented a cumulative of 6 predictive components calculated by the model and validated by a permutation (n = 10) (p = 0.05).

this matrix had the most comprehensive set of samples across years, and bacteria inhabiting stream sediments act as better proxies for longer term stream impacts[42]. A directional Principal Cooridinates Analysis (PCoA) plot generated using weighted Unifrac distances at the genus level displayed a decrease in variation among samples as the number of active wells (Fig. 3A) and wellpads (Fig. 3B) increased. Samples with zero or few active wells exhibited variable community structure as signified by the spread of samples along PC1 (Unifrac distance: 9.702) (Fig. 3). Samples with many active wells in the watershed were significantly less variable with a Unifrac distance of 0.0569 between the sample cluster and appeared to have greater similarity between samples at the genus level (Fig. 3). Further, a relationship between increased stream water acidity and bacterial community structure as a function of the count of active wells nearby the stream suggested that fracking activity may be selecting for specific bacterial assemblages within impacted stream environments.

Identification of Microbial Indicators. Biomarker analyses revealed an enrichment of specific OTUs within MSA+ and MSA− sediments, suggesting that certain OTUs could be readily responding to environmental perturbations (Fig. 4). In MSA− streams, members of Gemmatimonadetes were >3 $\log_{10}$-fold more abundant and *Myxococcus*, *Rhodobacter*, and Sphingomonadales members were >2 $\log_{10}$-fold more abundant. Methanobacteriaceae, Methylocystaceae, Beijerinckiaceae, Caulobacteraceae, and members of Pedosparaea were amongst the most significantly enriched in MSA+ sediment samples (Fig. 4). Specifically, both *Methanobacterium* and *Beijerinckia* were >2 $\log_{10}$-fold more abundant in MSA+ streams. Specific taxa were consistently enriched across many sampling years when biomarker analyses were completed with OTU relative abundances filtered by year to account for possible pseudoreplication (Fig. S6). For example, *Methanobacterium* sequences were enriched in MSA+ sites from 2012–2014 while Beijerinckiaceae sequences were increasingly present in later samples (2014–2015). Members of Verrucomicrobia, Sphingomonadales, and Myxococcaceae were consistently enriched in MSA− sites each year. Further, many of these bacteria also shared strong positive relationships with increased number of wellpads within the watershed, including taxa within the Caulobacteraceae, Methanobacteriaceae, Acetobacteraceae, and Methylocystaceae families (FDR-adjusted $p < 0.05$).

A survey of *Methanobacterium* abundance in sediment samples across all stream sites confirmed that the enrichment of these archaeal taxa was not singular to one stream site. Several MSA+ sites including both documented spill sites Alex Branch (ALXS) and Little Laurel Run (LiLRS) had the distinctly higher abundances of *Methanobacterium* each year (Fig. 5). *Methanobacterium* accounted for 0.14% of sequences in Alex Branch and 0.13% in Little Laurel in 2013. Indian Run (HRS) also exhibited significantly higher abundance of *Methanobacterium* in 2015 (0.11%), during which there were a total of 12 active wells within the watershed. Many other stream sites consistently harbored *Methanobacterium*, both within MSA+ and MSA− sites; however, normalized abundances did not exceed 3 for MSA− sites (Fig. 5).

The apparent enrichment of different OTUs across all years between MSA+ and MSA− sites led us to investigate whether MSA status could be predicted by differential OTU abundance. PLS-DA analyses showed that active well count was the most significant measured environmental selector, which directly corresponded to active fracking. Therefore, samples were reassigned as fracked (HF+) and non-fracked (HF−) based on actively producing wells (rather than wellpad presence). A random forest model was performed with an OTU table combined

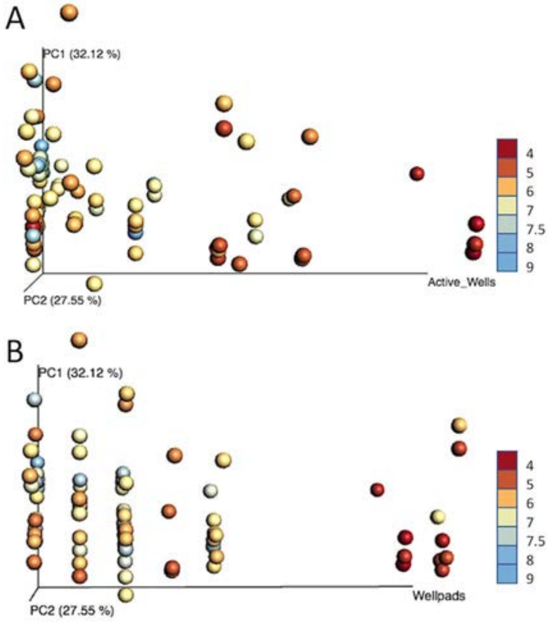

Figure 3. Directional Principal Coordinate Analysis (PCoA) plots of weighted Unifrac distances of sediment samples. The OTU table was CSS normalized and filtered to the genus level. Samples were plotted according to number of active wells (**A**) and wellpads (**B**) along the horizontal axis of the directional PCoA plot. Samples are colored by pH with high pH = blue, low pH = red. The horizontal axis represents from left to right, zero wells and wellpads to 21 active wells and 9 wellpads. Samples are closer in Unifrac distance as the number of active wells increase along PC1, suggesting less variation in bacterial community composition between samples.

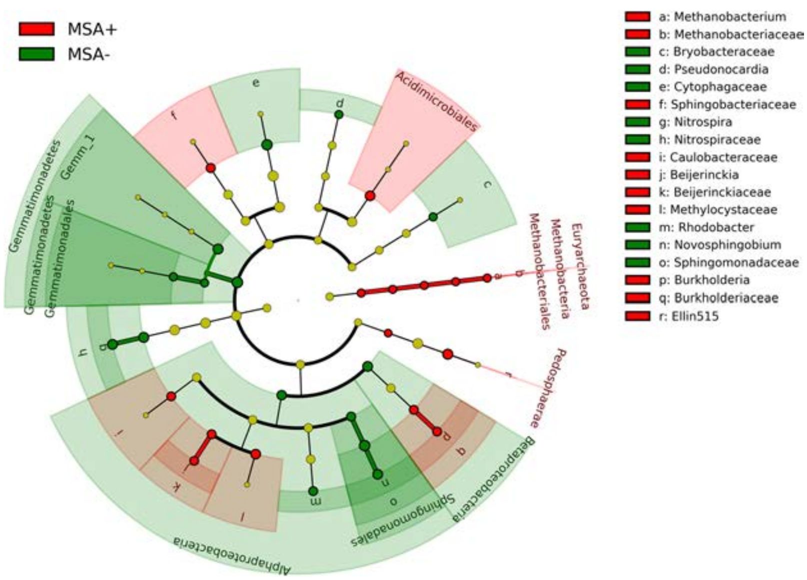

Figure 4. LEfSe plot for clades of bacteria enriched within MSA+ and MSA− streams. The cladogram reports the taxa (highlighted by small circles and shading (MSA+ = red; MSA− = green) that are enriched within corresponding sediment samples. LEfSe utilizes Kruskal-Wallis to determine significantly different taxonomic features ($p < 0.01$) between experimental groups, a pairwise Wilcoxon rank sum statistic to test biological consistency across subgroups ($p < 0.01$), and finally a linear discriminant analysis (LDA score >2.0) to determine the effect size, or magnitude of variation of the features between groups.

with measured environmental data, revealing that active wells, stream, wellpads, and pH were all significant predictors of fracking status. Random forest modelling was continued with only the OTU table to identify potential microbial predictors of active fracking activity based on differences in OTU abundances. Thirty OTUs with the largest mean decrease in Gini index were selected (Table S6). The three most significant predictors were Sphingobacteriales, Cytophagaceae, and Solibacterales, signifying they had a large difference in their abundances between HF+ and HF− sites and could consequently represent accurate predictors fracking status for a site. Of

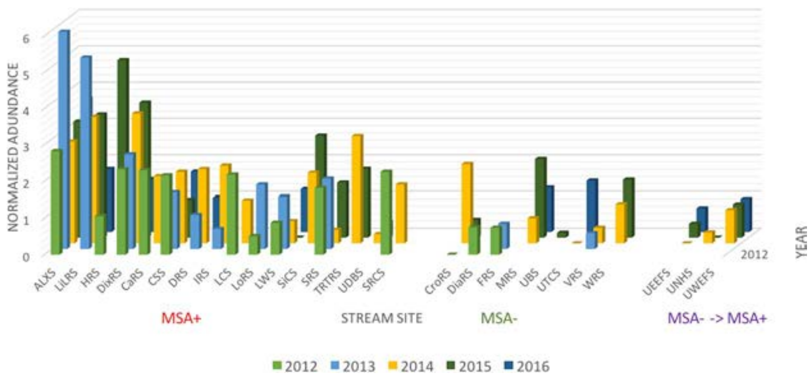

Figure 5. Relative abundance plot of *Methanobacterium* sequences in sediment samples (n = 86). Samples were grouped by MSA+, MSA−, and MSA− to MSA+ status. A filtered OTU table that underwent CSS normalization was used, showing stream site along the x-axis and year along the z-axis. *Methanobacterium* presence was stable across years and showed greater abundance in MSA+ sites.

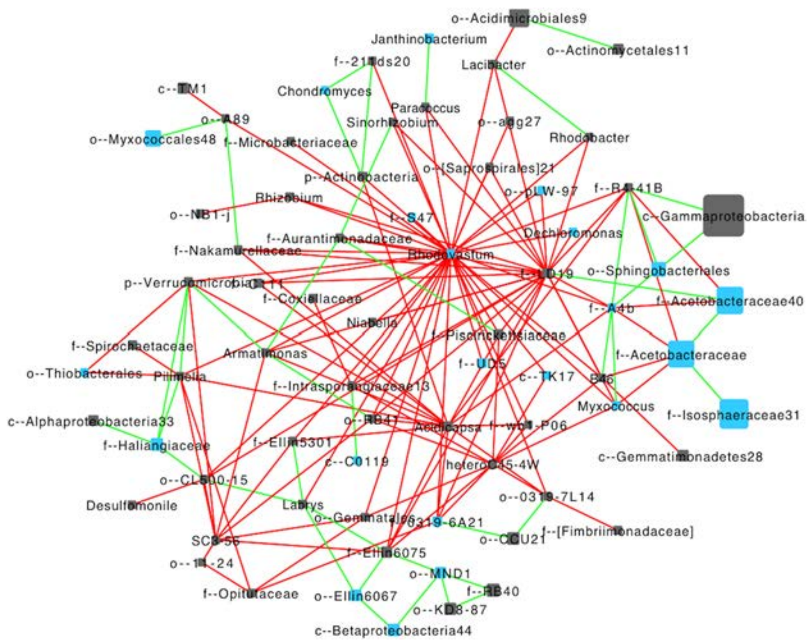

Figure 6. Co-occurrence network bacterial taxa within a selection of MSA+ (n = 14) stream sites. The network plot was generated within the Cytoscape plug-in Conet and reveals strong positive (Spearman's rho >0.8) and strong negative (Spearman's rho <−0.8) correlations. Edges connecting nodes highlighted in green are indicative of strong positive correlation, whereas edges highlighted in red are indicative of a strong negative correlation. Sizes of bacterial nodes indicate bacterial abundance. Taxa identified as important predictors are colored in blue.

the top thirty predictors identified by random forest, three were significantly enriched in MSA+ streams and twelve were significantly enriched in MSA− streams. The percentage classified correctly value for all 100 random forest models was calculated to be 0.686, indicating that the random forest models predicted fracking status at a percentage higher than expected by chance.

Co-occurrence network patterns of bacterial communities within both MSA− and MSA+ stream sediments displayed strong relationships. A subnetwork containing bacterial correlations within MSA+ streams revealed that there were strong positive and negative correlations between taxa that were previously identified as important predictors of nearby fracking activity (Fig. 6). For example, a positive relationship was revealed between several OTUs within the Acetobacteraceae family. Further, Acetobacteraceae shared the only positive correlation with the family LD19 within the Methylacidiphales order, which had 23 negative correlations with other OTUs. Gammaproteobacteria were the most abundant group, and shared positive correlations with taxa of the Sphingobacteriales order and family R4–41B within the Pedosphaerales order (Fig. 6). Sequences matching to *Rhodovastum*, a member of the Acetobacteraceae family, shared the most negative correlations (37) with other OTUs.

Discussion

Five years of stream water chemistry data indicated that stream sites located in watersheds with Marcellus Shale activity (MSA+) had lower pH than MSA− sites. While these differences could be partly attributed to a number of variants (i.e. watershed characteristics, disparities in acid rain deposition, or fracking activity), sites were selected using GIS surveys to avoid potential confounds[36]. Further, it has been shown that pH significantly differs in sites with and without fracking activity[36,40,43,44]. Observed differences in pH of stream water could be attributed to weathering of pyritic geological formations[45] exposed during the drilling process or exposure to concentrated acids that are used within fracking fluids during the hydraulic fracturing process[33,40]. A negative correlation between the number of active wells and pH suggests that activity associated with actively producing wells could be increasing the acidity of stream ecosystems. Indeed, streams with the largest number of active wells, including the two documented spill sites, had significantly lower pH levels. It is important to note that stream sites selection excluded streams impacted from acid mine drainage. No other measured water chemistry measurements were significantly different, suggesting that pH differences were not related to differences in the sites that were selected but rather the effects of activity.

There are a few modes in which fracking could be shaping bacterial community dynamics. Numerous studies have previously documented apparent shifts in bacterial community resulting from changes in stream water pH[46–48]. Fracking activity could be causing differences in stream pH and thereby causing shifts of ecosystem dynamics[49] and communities[40,43,44]. It is also possible that aquatic bacteria are responding directly to fracking inputs. While alpha diversity was not indicative of extensive trends between MSA+ and MSA− streams (Table S5), the potential inputs from fracking-associated fluids during active fracking may not be frequent enough or encompass a large enough volume to cause a drastic restructuring of the aquatic bacterial communities. It is clear, however, that spills associated with fluid mishandling and other operational accidents immediately impacted nearby surface waters[50]. While older and inactive wells remain a critical concern for environmental contamination associated with equipment failure[51], operating wells are associated with increased truck transportation and fluid relocation, which could pose larger potential threats to the surrounding environment. PA treats most of its fracking wastewater off site, requiring extensive relocation of fluids following the fracking process[33]. Injection and flowback fluids are documented to have pH range of 6–8, but the concentration of acids (and bases) varies widely by company and well site[52].

Significant differences in beta diversity were observed at the family and genus-level (Fig. 3). As active wells increased, Unifrac distances between samples decreased, suggesting a shaping of the sediment bacterial communities (Fig. 3). Samples with zero active wells exhibited large variation in comparison to samples with as many as 21 active wells. This disparity in bacterial variation could be resulting from differential enrichment of bacterial assemblages that vary in their sensitivity to changes in stream conditions[53]. Previous studies investigating fracking impact on nearby aquatic systems have reported increased methanotrophic bacteria in impacted sites[31,40]. Specifically, in our study, members of the Caulobacteraceae, Beijerinckiaceae, Methylosystaceae, and Burkolderiaceeae were significantly enriched in MSA+ sites (Fig. 4). Caulobacteraceae, Beijerinckiaceae, and Methylosystaceae are known to contain bacteria with methanotrophic capabilities[54]. Further, Beijerinckiaceae and Methylocystaceae are also commonly found in acidic habitats, and they often prevail in methane-emitting wetlands[55,56]. Both Caulobacteraceae and Burkolderiaceae are composed of taxa with great metabolic versatility, as they can degrade a wide range of hydrocarbons[57].

Interestingly, among the bacteria most enriched in MSA+ stream sites were *Phenylobacterium*, which may be capable of degrading phenyl-compounds and other complex hydrocarbons in acidic environments[58,59]. Random forest analysis revealed that *Phenylobacterium* as well as many of the enriched biomarkers were important predictors of fracking activity. Moreover, *Rhodovastum*, a member of the Acetobacteraceae family, appeared to be a hallmark of MSA+ stream sites, indicated by its high degree of negative associations with other taxa (Fig. 6). *Rhodovastum* has not been extensively studied but has been isolated from methane-rich paddy soils[60] and is a putative hydrocarbon-degrader within methane-emitting fen soil[61]. The large degree of biological interactions within MSA+ sediment soil suggests that *Rhodovastum* could be among those participating in degrading hydrocarbon constituents of potential fracking inputs. Bacterial genera known to degrade aliphatic and aromatic compounds have been previously identified in fracking-associated fluids[37,39,62].

Enrichments in MSA− sites were comprised of several pH sensitive taxa, which were potentially repressed in MSA+ stream conditions. Gemmatimonadales favor neutral pH and have been found to decrease in abundance in more acidic conditions[63]. Additionally, Nitrospira is found in greater abundance in higher pH within soils[47], which is consistent with their enrichment in MSA− sites. Furthermore, certain biomarker taxa remained significantly enriched across individual sampling years. From 2012 to 2015, these biomarker taxa were repeatedly observed at significantly higher abundance when fracking activity was most prevalent across northwestern PA. In 2016, the total number of active wells across sample sites dropped to 37 from 116 in the previous year. Samples in 2016 contained different bacterial enrichments compared with previous years' (Fig. S6).

16S rRNA gene sequencing revealed that methanogenic taxa were significantly enriched in MSA+ sites during the five-year period. Sequences belonging to archaeal taxa were more abundant in MSA+ sites, including spill sites, and were dominated *Methanobacterium* (Fig. 5). This finding is consistent with a study that found downstream sites of unconventional oil and gas wastewater releases were dominated by *Methanomicrobia*, also methanogenic bacteria, which was attributed to changes in stream geochemistry following fracking wastewater inputs[31]. These observed differences are characteristic of unaerated and biocide-amended impoundments of produced water from unconventional oil and gas drilling[64], suggesting that MSA+ streams may share similar bacterial assemblages as those that have endured fracking-related spills. Methanogens are tolerant of acidic conditions, enabling them to survive in such environments[65].

The concomitant enrichment of both methanogenic and methanotrophic bacteria suggests that they may be co-occurring due to stream conditions. The increased acidity in MSA+ streams has potentially enriched methanogens[66], and the greater availability of both biogenic methane and potential methane emission from fracking has cultivated an enrichment of methanotrophic bacteria. Methanogenic and methanotropic bacteria have been shown to coexist in sediment habitats[67], but it remains inconclusive whether methane contamination is frequent enough nearby fracking activity to amplify this relationship. Stream sediments naturally contain methane and its intermediate constituents, and therefore, harbor methanogenic archaea, essential for methane production and consumption within the freshwater ecosystem[68,69]. However, subsets of groundwater wells <1 km from shale gas wells in PA have documented elevated concentrations of methane, ethane, propane, and methane isotopic signatures consistent with a thermogenic source, suggesting potential methane migration from fracking[70–72]. Conversely, it is reported that groundwater in northwestern, VA was not contaminated from the installation and fracking of shale-gas wells over the course of three years[50].

While high-throughput sequencing of the 16 S rRNA gene enabled us to identify bacteria responsive to proximate fracking activity, this approach should be employed with caution, due to the limited phylogenetic resolution when using the 16 S rRNA gene as a target. Future work such as shotgun metagenomics and metatranscriptomics could be used to investigate the functional response of microbial communities towards potential environmental disturbances associated with Marcellus shale activities. Future work should also focus on additional chemical measurements within stream water such as methane concentration and isotopic carbon to further connect abiotic conditions to microbial response, a current limit of this study. Future studies will expand to other areas of PA with greater presence and documented impacts Marcellus shale activity to investigate the spatial stability of in-stream biomarker taxa identified in this study.

Altogether, this study highlighted stable bacterial taxa responding to Marcellus shale activity and further supplements a longitudinal correlation of increased acidity of stream water and fracking activity adjacent to headwater streams over five years. While overall bacterial community composition did not show large-scale differences between MSA+ and MSA− streams, our results suggest that fracking activity may still be shaping community dynamics of select bacterial assemblages. These findings are relevant, as small headwater streams may be most impacted by the disruption associated with fracking operations, and these *in situ* bacterial communities comprise the first biological response. Understanding the dynamics of these aquatic bacterial communities and their potential capabilities will assist in attenuation of impacted sites and further inform environmental agendas associated with fracking operations.

Methods

Site Selection. All streams selected for sampling were located on public lands with necessary permits acquired through the Department of Conservation and Natural Resources (http://www.dcnr.state.pa.us) and the Pennsylvania Game Commission, SFRA-1322. All permits are available upon request.

Thirty-one Pennsylvanian headwater streams with unconventional shale gas well permits from the PA DEP were selected based on previously outlined criteria[44]. The selected streams were remotely located within the Marcellus shale region in northwestern PA within forested watersheds that had little to no prior anthropogenic activity. Further, the streams contained naturally reproducing wild brook trout populations. All sites shared similar watershed and stream characteristics to allow comparison with respect to the effect of Marcellus shale activity and fracking[44]. Sampling locations are displayed in Fig. 1.

Streams without fracking infrastructure development for the duration of the sampling period (2012–2016) were classified as lacking Marcellus shale activity or MSA− (n = 9). It is important to note that Dutch Hollow within the MSA− grouping had land cleared and roads constructed prior to sampling. Streams with at least one wellpad were categorized as MSA+ (n = 19). A wellpad was defined as land cleared for drilling operations that is occupied by a minimum of one well. Well presence on a wellpad was confirmed by the spud date or the date on which the ground was penetrated to drill the well. Because Alex Branch and UNT to Trout Run were both MSA+ streams and the sole inputs to Trout Run, Trout Run was categorized as MSA+ as well. Six streams changed from MSA− to MSA+ between 2012–2016 and were grouped separately. Two MSA+ streams (Little Laurel Run and Alex Branch) had documented fracking-associated contamination within the watershed before sampling began in 2012 according to the PADEP. For additional analyses, stream sites were also categorized as fracked (HF+) and non-fracked (HF−) based on actively producing wells within the watershed. Active wells within a watershed were defined as operating wells that were producing fluids for natural gas extraction.

Detailed information of selected watershed characteristics and impact status of each site over 5 years can be found in Supplemental Information (Table S1). It should be noted that not all sites were sampled every year because of high flow and/or inaccessibility.

Field Sampling. Sediment, water, bryophyte, and biofilm samples were collected over 5 years (2012–2016) in the summer months of June and July. Methods for sterile sample collection were utilized as described in Trexler *et al*. (2014). Sediment samples (n = 86) were collected using sterile scoops from areas adjacent to the water-bank interface. Biofilm samples (n = 5) were collected in sterile 50 mL conical tubes. Bryophyte samples (n = 20) were cut directly from submerged rock substrates with a sterile scalpel and consisted of two common water mosses, *Fontinalis sphagnifolia* and *Fontinalis antipyretica*. It should be noted that bryophyte samples were only collected in 2012 and 2013. Water samples (n = 78) were sampled by collecting 1 liter of water in a sterile Nalgene bottle from a central riffle. Water samples were filtered on site with 0.22 μm polyethersulfone filters (Millipore, Billerica, MA) and stored in sterile Whirl-Pak bags (Nasco, Fort Atkinson, WI). All samples were placed immediately on ice and stored at −80 °C. Stream water chemistry measurements including: temperature, pH, conductivity, salinity, and total dissolved solids (TDS) were taken on site at the time of sampling with a weekly-calibrated PCSTestr 35 (Oakton Instruments, Vernon Hills, IL).

DNA Extraction and 16 S rRNA library preparation. Nucleic acid extractions were performed on water filters, sediment, bryophyte, and biofilm samples using a modified cetyltrimethylammonium bromide (CTAB) phenol-chloroform-isoamyl alcohol method, as described by Hazen *et al.*[73]. The pellet was resuspended in 30 µL buffer EB (Qiagen, Germantown, MD) and the DNA was then subsequently subjected to the AllPrep DNA/RNA Mini Kit (Qiagen, Germantown, MD), using the manufacturer's suggested protocol. The DNA extracts were quantified using the Qubit 2.0 fluorometer double-stranded DNA (dsDNA) high sensitivity DNA kit (Invitrogen, Carlsbad, CA) according to the manufacturer's instructions and stored at −20 °C.

Duplicate 25 µL Illumina tag Polymerase Chain Reactions (PCR) from each sample ($n = 281$) contained final concentrations of 1× PCR buffer, 0.8 mM dinucleoside triphosphates (dNTPs), 0.625 U of *Taq* polymerase, 0.4 µM 515 F forward primer, 0.4 µM Illumina 806 R reverse barcoded primer, and ~10 ng of template DNA per reaction. Sediment DNA extracts, in many cases, were diluted by 1:10 in DEPC-treated water to achieve successful amplification. PCR was performed on an MJ Research PTC-200 thermocycler (Bio-Rad, Hercules, CA) using cycling conditions of 94 °C for 3 min, followed 35 cycles of 94 °C for 45 s, 53 °C for 60 s, and 72 °C for 90 s, and ending with 72 °C for 10 min. PCR reactions were kept at 4 °C until visualized on a 2% agarose E-gel (Invitrogen, Carlsbad, CA) stained with ethidium bromide. Pooled PCR products were gel purified using the Qiagen Gel Purification Kit (Qiagen, Frederick, MD), quantified using the Qubit 2.0 Fluorometer (Life Technologies, Carlsbad, CA), and validated using the Agilent Bioanalyzer High Sensitivity DNA kit (Agilent Technologies, Santa Clara, CA, USA) prior to submission for sequencing.

Sequencing. Due to the temporal nature of this study, libraries were sequenced by different facilities depending on the year. 2012 samples were sequenced using Illumina MiSeq set for 250 bp paired-end chemistry[40]. 2013 samples were sequenced through the EMP Consortium on the Illumina HiSeq. 2000 platform using the single-end 100 bp chemistry. Finally, libraries from 2014–2016 were sequenced on the Illumina Miseq platform with either the 150 bp or 250 bp paired-end chemistries. To normalize for different sequencing runs, we only used the first 100 bp of the forward reads. Quality control stringency and same-length truncation produced no significant run-to-run variation. Libraries from 2012 to 2016 were compiled into one multiplexed file for downstream sequence analysis. Sequence data for this project were deposited in the NCBI Sequence Read Archive under accession number SRP114850 (http://www.ncbi.nlm.nih.gov/sra).

Bioinformatics and Statistical Analyses. Sequence reads were trimmed at a length of 100 base pairs to normalize for different sequence lengths and quality filtered at an expected error of less 0.5% using USEARCH v7[74]. After quality filtering, the reads were analyzed using QIIME 1.9.0[75]. Open reference operational taxonomic units (OTUs) were picked and chimeric sequences were identified using USEARCH7[76]. OTU taxonomy was assigned using Greengenes 16 S rRNA database (13–5 release, 97%)[77]. The OTU table was filtered further to discard samples with less than 5400 sequences to remove samples with low sequencing depth because some diversity estimators can be sensitive to varying sample sizes. An additional filtering step was performed to discard OTUs that represented less the 0.005% sequences as recommended for Illumina-generated sequence data[78]. A total of 18.1 million sequences represented 189 samples (see Table S2 for filtering information for each sample).

Alpha-diversity multiple rarefactions were conducted using QIIME 1.9.0 on sequences across all samples from all years (2012–2016). To generate alpha diversity rarefaction curves multiple rarefactions were conducted on sequences across all samples from minimum depth of 500 sequences, to a maximum depth of 10,000 sequences, with a step size of 500 sequences/sample for 10 iterations. Alpha rarefactions were then collated and plotted using Chao1 and observed species richness metrics. Alpha diversity comparisons between sample matrix, stream status (MSA+ or MSA−), and measured water chemistry were conducted using two-sample t-test and nonparametric Monte Carlo permutations ($n = 999$). Visualization of trends in microbial community structure for MSA+ and MSA− samples were generated in R using the *phyloseq* package version 1.12.2[79].

Partial least square discriminant analysis (PLS-DA) was utilized to predict differences in sediments from streams with different numbers of proximal active wells. A cumulative sum scaling (CSS) normalized, unrarefied OTU table containing only sediment samples was used to create an OTU count matrix (X) and active well counts (Y) were used to predict maximal covariance of the samples. A six-component model was generated and validated by permutation ($n = 10$) ($p = 0.05$) to explain the relationship between the inputs (X variables) and the outputs (Y variables). Model quality was assessed by cross validation parameters R^2X, R^2Y, and Q^2. Cumulative R^2X and R^2Y values represent fraction of variance of the X and Y matrix while Q^2 represents the predictive accuracy of the model. A normalized OTU table was used to perform beta diversity analysis. Beta diversity was calculated using the weighted Unifrac distance metric and principal coordinate analysis (PCoA) plots were visualized using EMPeror[80]. Mean Unifrac distances were calculated between sample clusters were reported. Adonis tests were performed on the weighted Unifrac values to determine the variation explained by stream water characteristics, sample year, and watershed characteristics. All statistical analyses were considered significant at $\alpha = 0.05$ for both continuous and categorical variables.

Linear discriminant analysis (LDA) effect size (LEfSe) was utilized to identify specific bacterial taxonomic biomarkers unique to MSA− and MSA+ communities within sediment bacterial communities. The LEfSe method couples tests for statistical significance with other tests of effect relevance and biological consistency to determine the features, in this case OTUs, that most likely explain the differences in a phenotype or condition[81]. A normalized OTU table filtered to the genus-level and properly formatted[81]. OTU comparisons were performed with "impact status" (MSA+ or MSA−) set as the main categorical variable and "sample matrix" as the secondary categorical variable. Alpha levels of 0.01 were used for both the Kruskal-Wallis and pairwise Wilcoxon tests. Significant features with LDA >2 were plotted. The 'plot_cladogram' function was used to generate a cladogram for visualization of relationships between enriched bacterial taxa. To address the possibility of pseudoreplication in biomarker analyses, LEfSe was performed separately for each respective year (Fig. S6).

Statistical analysis of watershed characteristics (stream water pH, total dissolved solids (TDS), salinity, and temperature) as well as the number of active wells and wellpads were conducted between MSA+ and

MSA− streams using Wilcoxon rank sum test in R. Data was transformed ($\log_{10}$) and statistical significance was considered at $\alpha = 0.05$. Spearman correlations were calculated to examine the relationship between continuous abiotic variables of the stream sites.

Random forest modeling. The *randomForest* package in R[82] was implemented to produce a model that predicted fracking status of sediment samples using OTUs as predictors. The dataset consisted of un-normalized OTU counts with samples reclassified as fracked (HF+) or non-fracked (HF−). The initial step in the model generation was the separation of the full dataset into a training set and a test set, of which the training set consisted of a randomly selected subset containing approximately 80% of the samples and the test set consisted of the remaining 20% of the samples. Because no repeated samples existed within the dataset, the *sample* function was used to generate the randomly selected subset. The *randomForest* function was run on the training set, with the *mtry* parameter set to the square-root of the total number of predictors and all other default settings. To test the model, the *predict* function was run on the random forest model generated for the training set to predict the fracking status of the samples within the test set. The estimated test error was calculated as the total number of correct predictions divided by the total number of predictions made. The most important predictors were identified by implementing the *importance* function on the random forest model. The Mean Decrease in Gini Index was used as the measure of variable importance. The random forest model generation process was repeated 100 times so that the random forest model was run on 100 randomly selected training sets. An average estimated test error was calculated for all repetitions of the process. The top 30 predictors were assigned using the calculated average Mean Decrease in Gini Index of the 100 random forests produced.

Co-occurrence networks. Co-occurrence network plots generated in Cytoscape with the CoNet plug-in were used to investigate relationships between OTUs within MSA+ streams. A selection of 14 samples in streams representing all years (2012–2016) were used within the analysis. OTUs appearing in less than seven samples were discarded from the CSS normalized OTU table. Spearman correlation, Bray-Curtis dissimilarity and Kullback-Leibler dissimilarity were used to select edges. Thresholds for edge selection were chosen by CoNet such that the initial network would contain the 150 edges with the most positive correlations and the 150 edges with the most negative correlations for each of the four methods. The networks were then refined with a row-shuffling permutation step with 100 iterations. After permutation, the samples were renormalized. The permutation-renormalization step was intended to mitigate the compositionality of the data as described in Faust *et al.*[83]. The networks were further refined with a bootstrap step with 100 iterations. The networks created in the initial, permutation-renormalization, and bootstrap steps were combined using the Brown p-value merge and Benjamini-Hochberg multiple testing correction. The threshold for merged p-values was set at 0.05. Spearman rho thresholds were −0.82 and 0.85; thresholds of Bray-Curtis distances were 0.10 and 0.78.

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Author Contributions

R.L., N.U. and C.G. designed the study with input from M.C., T.H. and J.W.; N.U., C.G. and C.M. performed sample collection; N.U. analyzed the data with help from J.W.; V.K. generated the random forest model; R.D. created the co-occurrence networks; N.U. wrote the manuscript with R.L.

Additional Information

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ATTACHMENT C

STUDY 42

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Agency

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December 2016
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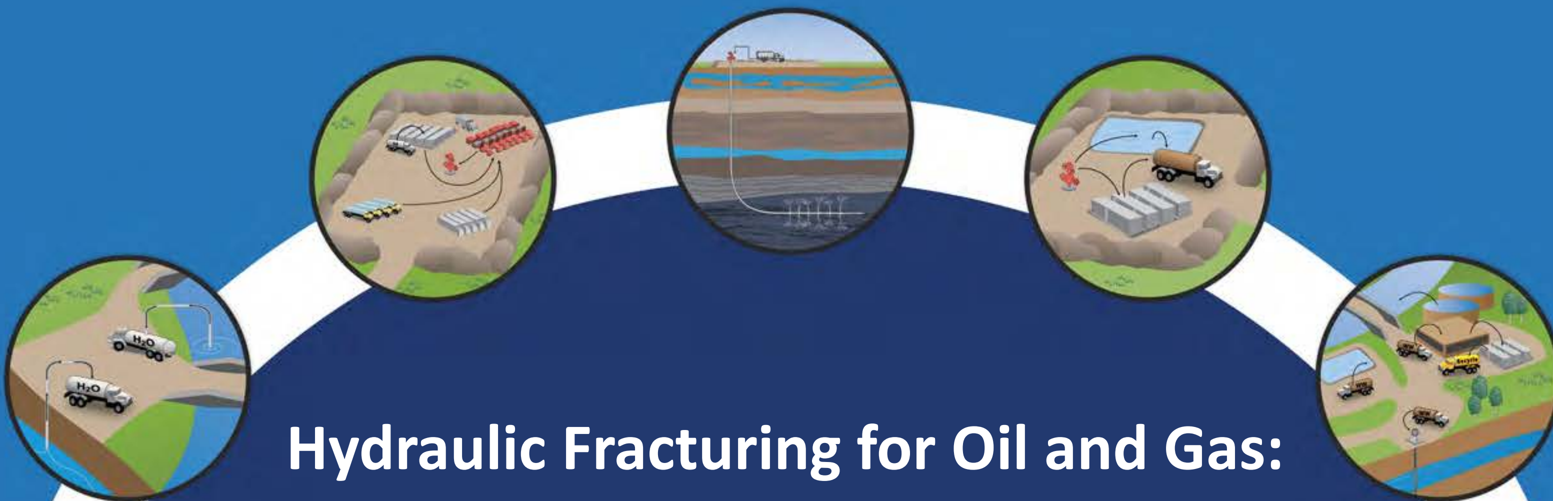

Hydraulic Fracturing for Oil and Gas: Impacts from the Hydraulic Fracturing Water Cycle on Drinking Water Resources in the United States

Executive Summary

Office of Research and Development
Washington, DC

Aerial photograph of hydraulic fracturing well sites near Williston, North Dakota.
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Executive Summary

People rely on clean and plentiful water resources to meet their basic needs, including drinking, bathing, and cooking. In the early 2000s, members of the public began to raise concerns about potential impacts on their drinking water from hydraulic fracturing at nearby oil and gas production wells. In response to these concerns, Congress urged the U.S. Environmental Protection Agency (EPA) to study the relationship between hydraulic fracturing for oil and gas and drinking water in the United States.

The goals of the study were to assess the potential for activities in the hydraulic fracturing water cycle to impact the quality or quantity of drinking water resources and to identify factors that affect the frequency or severity of those impacts. To achieve these goals, the EPA conducted independent research, engaged stakeholders through technical workshops and roundtables, and reviewed approximately 1,200 cited sources of data and information. The data and information gathered through these efforts served as the basis for this report, which represents the culmination of the EPA's study of the potential impacts of hydraulic fracturing for oil and gas on drinking water resources.

The hydraulic fracturing water cycle describes the use of water in hydraulic fracturing, from water withdrawals to make hydraulic fracturing fluids, through the mixing and injection of hydraulic fracturing fluids in oil and gas production wells, to the collection and disposal or reuse of produced water. These activities can impact drinking water resources under some circumstances. Impacts can range in frequency and severity, depending on the combination of hydraulic fracturing water cycle activities and local- or regional-scale factors. The following combinations of activities and factors are more likely than others to result in more frequent or more severe impacts:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;

- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;
- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

The above conclusions are based on cases of identified impacts and other data, information, and analyses presented in this report. Cases of impacts were identified for all stages of the hydraulic fracturing water cycle. Identified impacts generally occurred near hydraulically fractured oil and gas production wells and ranged in severity, from temporary changes in water quality to contamination that made private drinking water wells unusable.

The available data and information allowed us to qualitatively describe factors that affect the frequency or severity of impacts at the local level. However, significant data gaps and uncertainties in the available data prevented us from calculating or estimating the national frequency of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. The data gaps and uncertainties described in this report also precluded a full characterization of the severity of impacts.

The scientific information in this report can help inform decisions by federal, state, tribal, and local officials; industry; and communities. In the short-term, attention could be focused on the combinations of activities and factors outlined above. In the longer-term, attention could be focused on reducing the data gaps and uncertainties identified in this report. Through these efforts, current and future drinking water resources can be better protected in areas where hydraulic fracturing is occurring or being considered.

Drinking Water Resources in the United States

In this report, drinking water resources are defined as any water that now serves, or in the future could serve, as a source of drinking water for public or private use. This includes both surface water resources and groundwater resources (Text Box ES-1). In 2010, approximately 58% of the total volume of water withdrawn for public and non-public water supplies came from surface water resources and approximately 42% came from groundwater resources (Maupin et al., 2014).[1] Most people (86% of the population) in the United States relied on public water supplies for their drinking water in 2010, and approximately 14% of the population obtained drinking water from non-public water supplies. Non-public water supplies are often private water wells that supply drinking water to a residence.

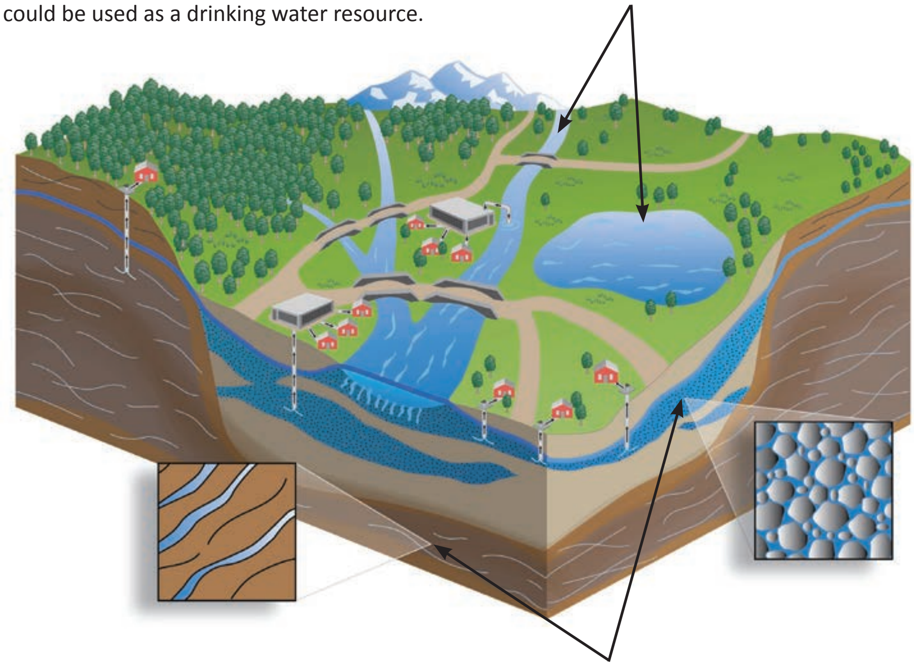
Future access to high-quality drinking water in the United States will likely be affected by changes in climate and water use. Since 2000, about 30% of the total area of the contiguous United States has experienced moderate drought conditions and about 20% has experienced severe drought conditions. Declines in surface water resources have

[1] Public water systems provide water for human consumption from surface or groundwater through pipes or other infrastructure to at least 15 service connections or serve an average of at least 25 people for at least 60 days a year. Non-public water systems have fewer than 15 service connections and serve fewer than 25 individuals.

Text Box ES-1: Drinking Water Resources

In this report, drinking water resources are considered to be any water that now serves, or in the future could serve, as a source of drinking water for public or private use. This includes both surface water bodies and underground rock formations that contain water.

Surface water resources include water bodies located on the surface of the Earth. Rivers, springs, lakes, and reservoirs are examples of surface water resources. Water quality and quantity are often considered when determining whether a surface water resource could be used as a drinking water resource.



Groundwater resources are underground rock formations that contain water. Groundwater resources are found at different depths nearly everywhere in the United States. Resource depth, water quality, and water yield are often considered when determining whether a groundwater resource could be used as a drinking water resource.

led to increased withdrawals and net depletions of groundwater in some areas. As a result, non-fresh water resources (e.g., wastewater from sewage treatment plants, brackish groundwater and surface water, and seawater) are increasingly treated and used to meet drinking water demand.

Natural processes and human activities can affect the quality and quantity of current and future drinking water resources. This report focuses on the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources; other processes or activities are not discussed.

Hydraulic Fracturing for Oil and Gas in the United States

Hydraulic fracturing is frequently used to enhance oil and gas production from underground rock formations and is one of many activities that occur during the life of an oil and gas production well (Figure ES-1). During hydraulic fracturing, hydraulic fracturing fluid is injected down an oil or gas production well and into the targeted rock formation under pressures great enough to fracture the oil- and gas-

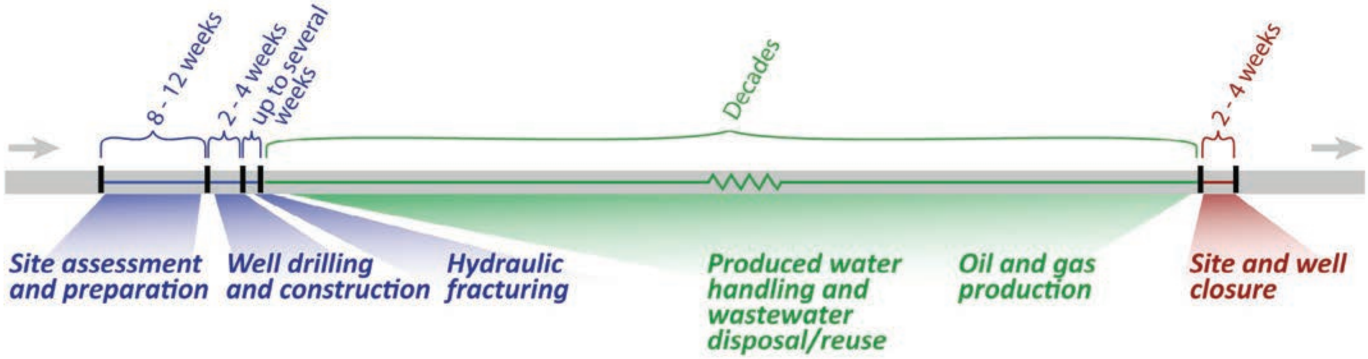


Figure ES-1. General timeline and summary of activities at a hydraulically fractured oil or gas production well.

bearing rock.[1] The hydraulic fracturing fluid usually carries proppant (typically sand) into the newly-created fractures to keep the fractures "propped" open. After hydraulic fracturing, oil, gas, and other fluids flow through the fractures and up the production well to the surface, where they are collected and managed.

Hydraulically fractured oil and gas production wells have significantly contributed to the surge in domestic oil and gas production, accounting for slightly more than 50% of oil production and nearly 70% of gas production in 2015 (EIA, 2016a, b). The surge occurred when hydraulic fracturing was combined with directional drilling technologies around 2000. Directional drilling allows oil and gas production wells to be drilled horizontally or directionally along the targeted rock formation, exposing more of the oil- or gas-bearing rock formation to the production well. When combined with directional drilling technologies, hydraulic fracturing expanded oil and gas production to oil- and gas-bearing rock formations previously considered uneconomical. Although hydraulic fracturing is commonly associated with oil and gas production from deep, horizontal wells drilled into shale (e.g., the Marcellus Shale in Pennsylvania or the Bakken Shale in North Dakota), it has been used in a variety of oil and gas production wells (Text Box ES-2) and other types of oil- or gas-bearing rock (e.g., sandstone, carbonate, and coal).

Approximately 1 million wells have been hydraulically fractured since the technique was first developed in the late 1940s (Gallegos and Varela, 2015; IOGCC, 2002). Roughly one third of those wells were hydraulically fractured between 2000 and approximately 2014. Wells hydraulically fractured between 2000 and 2013 were located in pockets of activity across the United States (Figure ES-2). Based on several different data compilations, we estimate that 25,000 to 30,000 new wells were drilled and hydraulically fractured in the United States each year between 2011 and 2014, in addition to existing wells that were hydraulically fractured to increase production.[2] Following the decline in oil and gas prices, the number of new wells drilled and hydraulically fractured appears to have decreased, with about 20,000 new wells drilled and hydraulically fractured in 2015.

Hydraulically fractured oil and gas production wells can be located near or within sources of drinking water. Between 2000 and 2013, approximately 3,900 public water systems were estimated to have had at least one hydraulically fractured well within 1 mile of their water source; these public water systems served more than 8.6 million people year-round in 2013. An additional 3.6 million people were estimated to have obtained drinking water from non-

[1] The targeted rock formation (sometimes called the "target zone" or "production zone") is the portion of a subsurface rock formation that contains the oil or gas to be extracted.

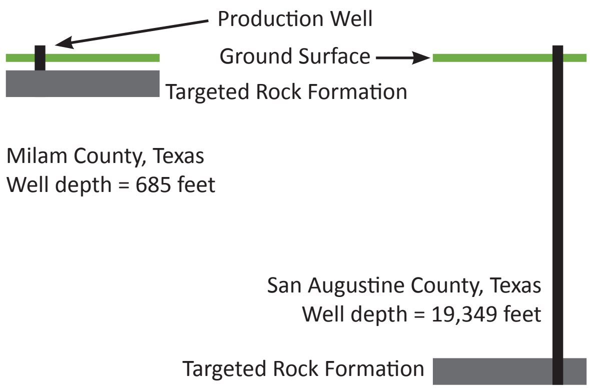
[2] See Table 3-1 in Chapter 3.

Text Box ES-2: Hydraulically Fractured Oil and Gas Production Wells

Hydraulically fractured oil and gas production wells come in different shapes and sizes. They can have different depths, orientations, and construction characteristics. They can include new wells (i.e., wells that are hydraulically fractured soon after construction) and old wells (i.e., wells that are hydraulically fractured after producing oil and gas for some time).

Well Depth

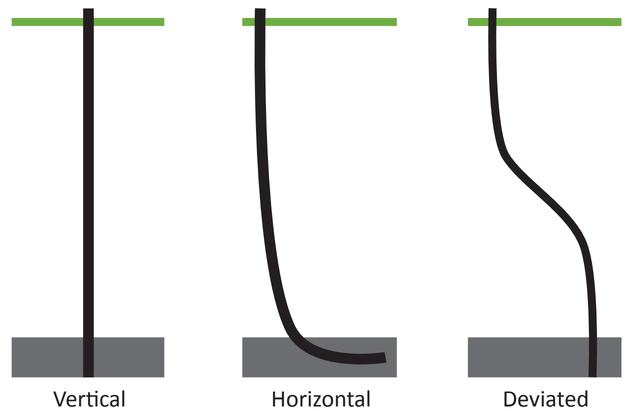
Wells can be relatively shallow or relatively deep, depending on the depth of the targeted rock formation.



Well depths and locations from FracFocus.org.

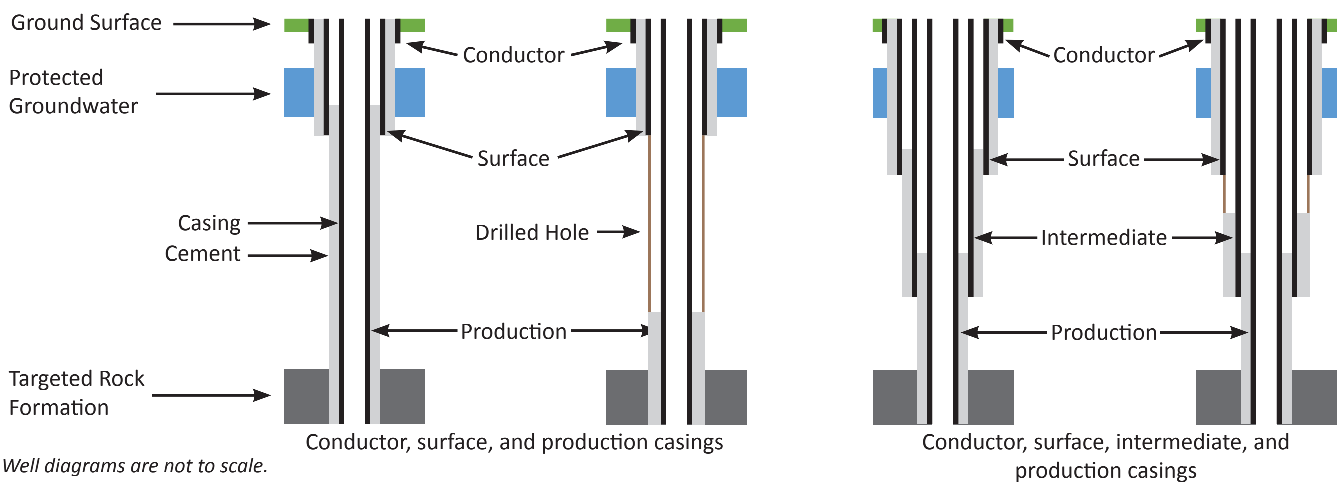
Well Orientation

Wells can be vertical, horizontal, or deviated.



Well Construction Characteristics

Wells are typically constructed using multiple layers of casing and cement. The subsurface environment, state and federal regulations, and industry experience and practices influence the number and placement of casing and cement.



Well diagrams are not to scale.

Oil and Gas Production Well Dictionary

Casing	Steel pipe that extends from the ground surface to the bottom of the drilled hole
Cement	A slurry that hardens around the outside of the casing; cement fills the space between casings or between a casing and the drilled hole and provides support for the casing
Conductor casing	Casing that prevents the in-fill of dirt and rock in the uppermost few feet of drilled hole
Intermediate casing	Casing that seals off intermediate rock formations that may have different pressures than deeper or shallower rock formations
Production casing	Casing that transports fluids up and down the well
Surface casing	Casing that seals off groundwater resources that are identified as drinking water or useable
Targeted rock formation	The part of a rock formation that contains the oil and/or gas to be extracted

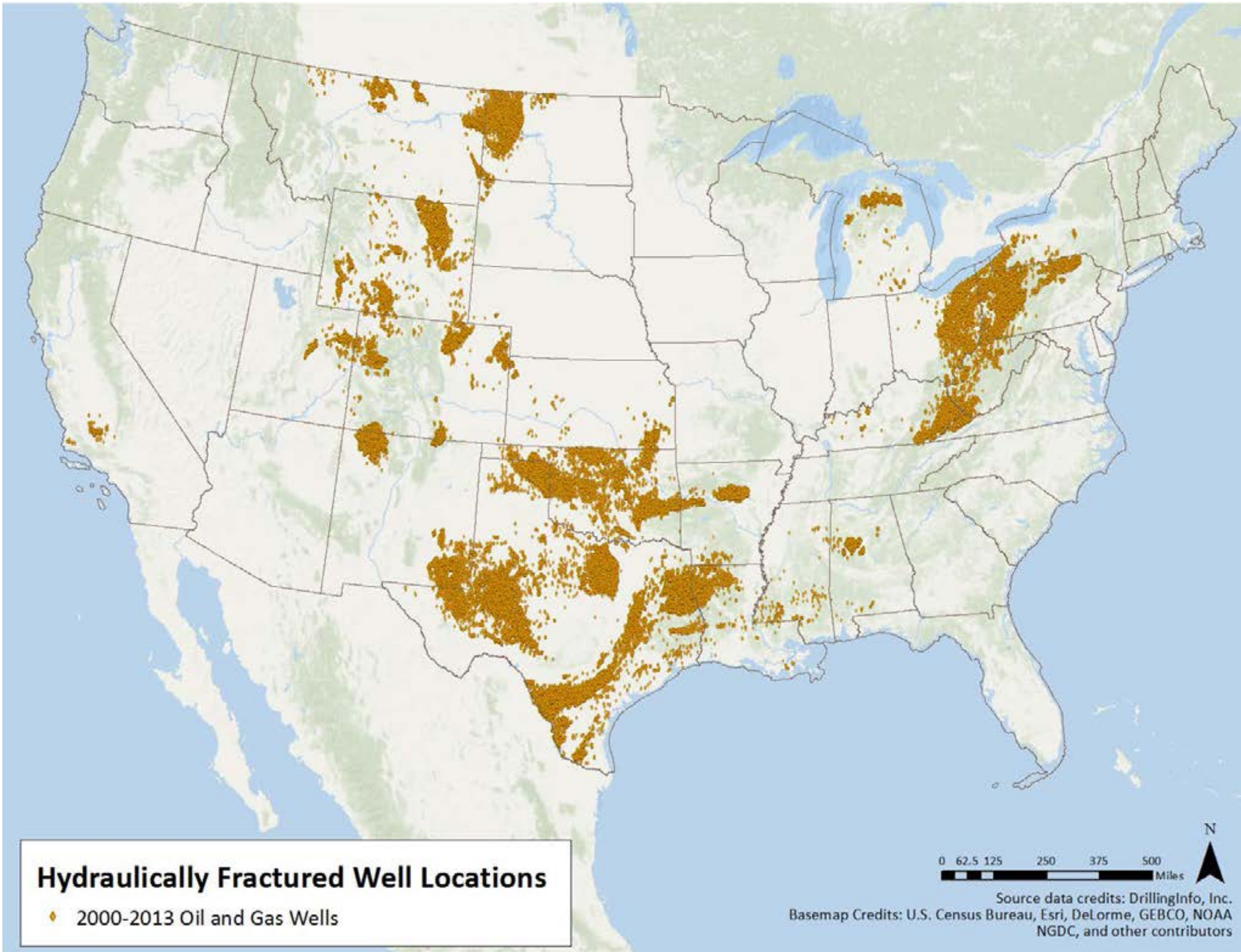


Figure ES-2. Locations of approximately 275,000 wells that were drilled and likely hydraulically fractured between 2000 and 2013. Data from DrillingInfo (2014).

public water supplies in counties with at least one hydraulically fractured well.[1] Underground, hydraulic fracturing can occur in close vertical proximity to drinking water resources. In some parts of the United States (e.g., the Powder River Basin in Montana and Wyoming), there is no vertical distance between the top of the hydraulically fractured oil- or gas-bearing rock formation and the bottom of treatable water, as determined by data from state oil and gas agencies and state geological survey data.[2] In other parts of the country (e.g., the Eagle Ford Shale in Texas), there can be thousands of feet of rock that separate treatable water from the hydraulically fractured oil- or gas-bearing rock formation. When hydraulically fractured oil and gas production wells are located near or within drinking water resources, there is a greater potential for activities in the hydraulic fracturing water cycle to impact those resources.

[1] This estimate only includes counties in which 30% or more of the population (i.e., two or more times the national average) relied on non-public water supplies in 2010. See Section 2.5 in Chapter 2.

[2] In these cases, water that is naturally found in the oil- and gas-bearing rock formation meets the definition of drinking water in some parts of the basin. See Section 6.3.2 in Chapter 6.

Approach: The Hydraulic Fracturing Water Cycle

The EPA studied the relationship between hydraulic fracturing for oil and gas and drinking water resources using the hydraulic fracturing water cycle (Figure ES-3). The hydraulic fracturing water cycle has five stages; each stage is defined by an activity involving water that supports hydraulic fracturing. The stages and activities of the hydraulic fracturing water cycle include:

- **Water Acquisition:** the withdrawal of groundwater or surface water to make hydraulic fracturing fluids;
- **Chemical Mixing:** the mixing of a base fluid (typically water), proppant, and additives at the well site to create hydraulic fracturing fluids;[1]
- **Well Injection:** the injection and movement of hydraulic fracturing fluids through the oil and gas production well and in the targeted rock formation;
- **Produced Water Handling:** the on-site collection and handling of water that returns to the surface after hydraulic fracturing and the transportation of that water for disposal or reuse;[2] and
- **Wastewater Disposal and Reuse:** the disposal and reuse of hydraulic fracturing wastewater.[3]

Potential impacts on drinking water resources from the above activities are considered in this report. We do not address other concerns that have been raised by stakeholders about hydraulic fracturing (e.g., potential air quality impacts or induced seismicity) or other oil and gas exploration and production activities (e.g., environmental impacts from site selection and development), as these were not included in the scope of the study. Additionally, this report is not a human health risk assessment; it does not identify populations exposed to hydraulic fracturing-related chemicals, and it does not estimate the extent of exposure or estimate the incidence of human health impacts.

Each stage of the hydraulic fracturing water cycle was assessed to identify (1) the potential for impacts on drinking water resources and (2) factors that affect the frequency or severity of impacts. Specific definitions used in this report are provided below:

- An **impact** is any change in the quality or quantity of drinking water resources, regardless of severity, that results from an activity in the hydraulic fracturing water cycle.
- A **factor** is a feature of hydraulic fracturing operations or an environmental condition that affects the frequency or severity of impacts.
- **Frequency** is the number of impacts per a given unit (e.g., geographic area, unit of time, number of hydraulically fractured wells, or number of water bodies).
- **Severity** is the magnitude of change in the quality or quantity of a drinking water resource as measured by a given metric (e.g., duration, spatial extent, or contaminant concentration).

[1] A base fluid is the fluid into which proppants and additives are mixed to make a hydraulic fracturing fluid; water is an example of a base fluid. Additives are chemicals or mixtures of chemicals that are added to the base fluid to change its properties.

[2] "Produced water" is defined in this report as water that flows from and through oil and gas wells to the surface as a byproduct of oil and gas production.

[3] "Hydraulic fracturing wastewater" is defined in this report as produced water from hydraulically fractured oil and gas wells that is being managed using practices that include, but are not limited to, injection in Class II wells, reuse in other hydraulic fracturing operations, and various aboveground disposal practices. The term "wastewater" is being used as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. Class II wells are used to inject wastewater associated with oil and gas production underground and are regulated under the Underground Injection Control Program of the Safe Drinking Water Act.

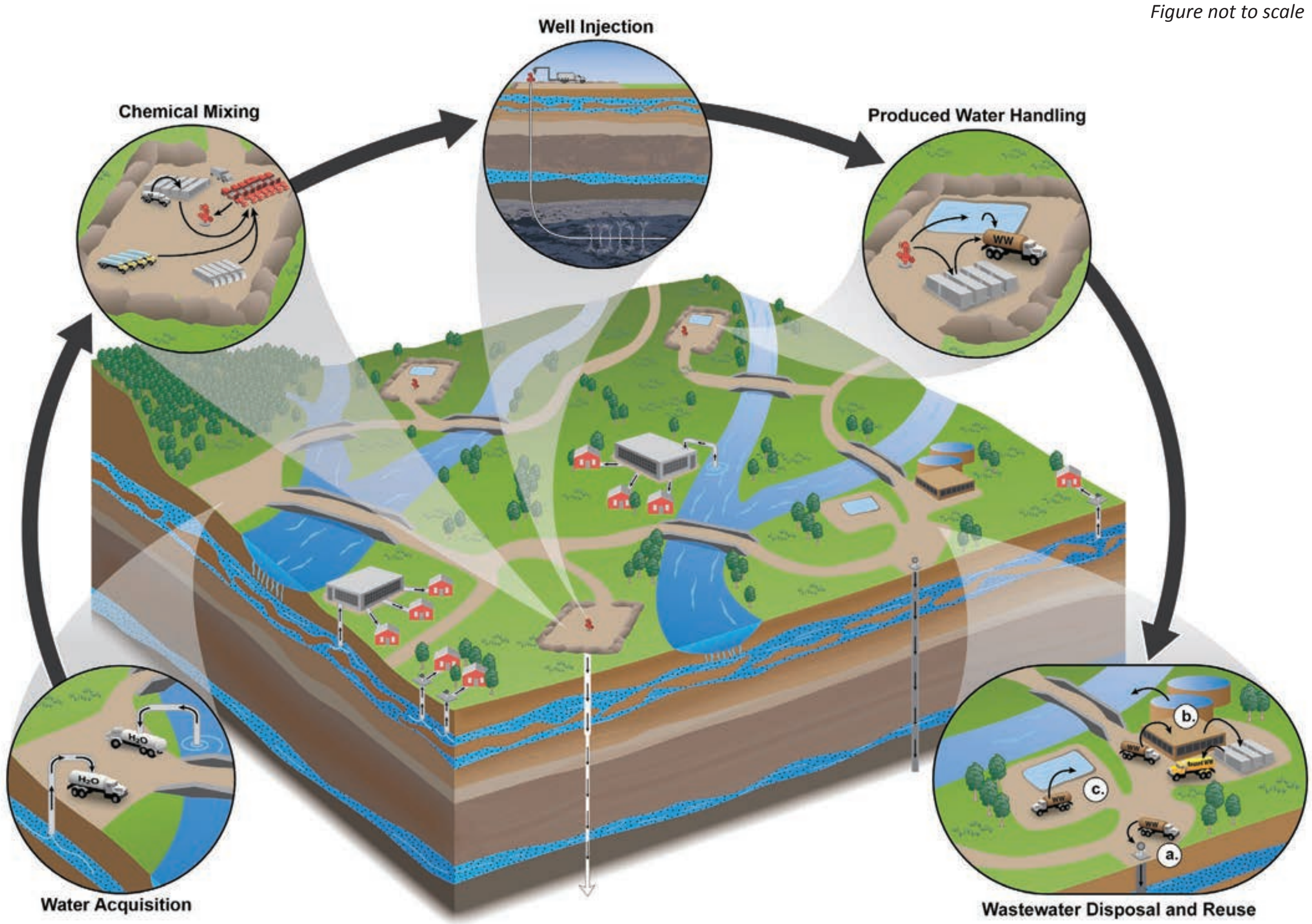


Figure ES-3. The five stages of the hydraulic fracturing water cycle. The stages (shown in the insets) identify activities involving water that support hydraulic fracturing for oil and gas. Activities may take place in the same watershed or different watersheds and close to or far from drinking water resources. Thin arrows in the insets depict the movement of water and chemicals. Specific activities in the "Wastewater Disposal and Reuse" inset include (a) disposal of wastewater through underground injection, (b) wastewater treatment followed by reuse in other hydraulic fracturing operations or discharge to surface waters, and (c) disposal through evaporation or percolation pits.

Factors affecting the frequency or severity of impacts were identified because they describe conditions under which impacts are more or less likely to occur and because they could inform the development of future strategies and actions to prevent or reduce impacts. Although no attempt was made to identify or evaluate best practices, ways to reduce the frequency or severity of impacts from activities in the hydraulic fracturing water cycle are described in this report when they were reported in the scientific literature. Laws, regulations, and policies also exist to protect drinking water resources, but a comprehensive summary and broad evaluation of current or proposed regulations and policies was beyond the scope of this report.

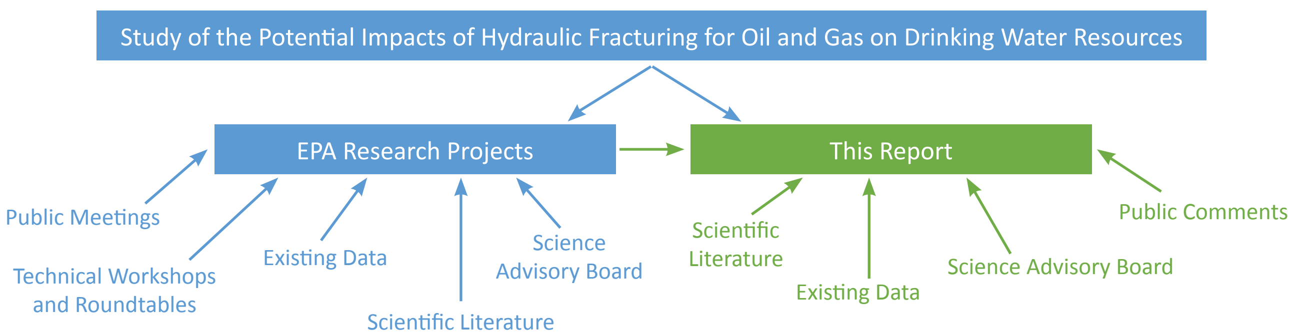
Relevant scientific literature and data were evaluated for each stage of the hydraulic fracturing water cycle. Literature included articles published in science and engineering journals, federal and state government reports, non-governmental organization reports, and industry publications. Data sources included federal- and state-collected data sets, databases maintained by federal and state government agencies, other publicly available data, and industry data provided to the EPA.[1] The relevant literature and data complement research conducted by the EPA under its *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (Text Box ES-3).

A draft of this report underwent peer review by the EPA's Science Advisory Board (SAB). The SAB is an independent federal advisory committee that often conducts peer reviews of high-profile scientific matters relevant to the EPA. Members of the SAB and *ad hoc* panels formed under the auspices of the SAB are nominated by the public and selected based on factors such as technical expertise, knowledge, experience, and absence of any real or perceived conflicts of interest. Peer review comments provided by the SAB and public comments submitted to the SAB during their peer review, including comments on major conclusions and technical content, were carefully considered in the development of this final document.

A summary of the activities in the hydraulic fracturing water cycle and their potential to impact drinking water resources is provided below, including what is known about human health hazards associated with chemicals identified across all stages of the hydraulic fracturing water cycle. Additional details are available in the full report.

Text Box ES-3: The EPA's *Study of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources*

The EPA's study is the first national study of the potential impacts of hydraulic fracturing for oil and gas on drinking water resources. It included independent research projects conducted by EPA scientists and contractors and a state-of-the-science assessment of available data and information on the relationship between hydraulic fracturing and drinking water resources (i.e., this report).



Throughout the study, the EPA consulted with the Agency's independent Science Advisory Board (SAB) on the scope of the study and the progress made on the research projects. The SAB also conducted a peer review of both the *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2011; referred to as the *Study Plan* in this report) and a draft of this report.

Stakeholder engagement also played an important role in the development and implementation of the study. While developing the scope of the study, the EPA held public meetings to get input from stakeholders on the study scope and design. While conducting the study, the EPA requested information from the public and engaged with technical, subject-matter experts on topics relevant to the study in a series of technical workshops and roundtables. For more information on the EPA's study, including the role of the SAB and stakeholders, visit www.epa.gov/hfstudy.

[1] Industry data was provided to the EPA in response to two separate information requests to oil and gas service companies and oil and gas production well operators. Some of these data were claimed as confidential business information under the Toxic Substances Control Act and were treated as such in this report.

Water Acquisition

The withdrawal of groundwater or surface water to make hydraulic fracturing fluids.

Relationship to Drinking Water Resources

Groundwater and surface water resources that provide water for hydraulic fracturing fluids can also provide drinking water for public or non-public water supplies.

Water is the major component of nearly all hydraulic fracturing fluids, typically making up 90–97% of the total fluid volume injected into a well. The median volume of water used, per well, for hydraulic fracturing was approximately 1.5 million gallons (5.7 million liters) between January 2011 and February 2013, as reported in FracFocus 1.0 (Text Box ES-4). There was wide variation in the water volumes reported per well, with 10^{th} and 90^{th} percentiles of 74,000 gallons (280,000 liters) and 6 million gallons (23 million liters) per well, respectively. There was also variation in water use per well within and among states (Table ES-1). This variation likely results from several factors, including the type of well, the fracture design, and the type of hydraulic fracturing fluid used. An analysis of hydraulic fracturing fluid data from Gallegos et al. (2015) indicates that water volumes used per well have increased over time as more horizontal wells have been drilled.

Water used for hydraulic fracturing is typically fresh water taken from available groundwater and/or surface water resources located near hydraulically fractured oil and gas production wells. Water sources can vary across the United States, depending on regional or local water availability; laws, regulations, and policies; and water management practices. Hydraulic fracturing operations in the humid eastern United States generally rely on surface water

Text Box ES-4: FracFocus Chemical Disclosure Registry

The FracFocus Chemical Disclosure Registry is a publicly-accessible website (www.fracfocus.org) managed by the Ground Water Protection Council (GWPC) and the Interstate Oil and Gas Compact Commission (IOGCC). Oil and gas production well operators can disclose information at this website about water and chemicals used in hydraulic fracturing fluids at individual wells. In many states where oil and gas production occurs, well operators are required to disclose to FracFocus well-specific information on water and chemical use during hydraulic fracturing.

The GWPC and the IOGCC provided the EPA with over 39,000 PDF disclosures submitted by well operators to FracFocus (version 1.0) before March 1, 2013. Data in the disclosures were extracted and compiled in a project database, which was used to conduct analyses on water and chemical use for hydraulic fracturing. Analyses were conducted on over 38,000 unique disclosures for wells located in 20 states that were hydraulically fractured between January 1, 2011, and February 28, 2013.

Despite the challenge of adapting a dataset originally created for local use and single-PDF viewing to answer broader questions, the project database created by the EPA provided substantial insight into water and chemical use for hydraulic fracturing. The project database represents the data reported to FracFocus 1.0 rather than all hydraulic fracturing that occurred in the United States during the study time period. The project database is an incomplete picture of all hydraulic fracturing due to voluntary reporting in some states for certain time periods (in the absence of state reporting requirements), the omission of information on confidential chemicals from disclosures, and invalid or erroneous information in the original disclosures or created during the development of the database. The development of FracFocus 2.0, which became the exclusive reporting mechanism in June 2013, was intended to increase the quality, completeness, and consistency of the data submitted by providing dropdown menus, warning and error messages during submission, and automatic formatting of certain fields. The GWPC has announced additional changes and upgrades for FracFocus 3.0 to enhance data searchability, increase system security, provide greater data accuracy, and further increase data transparency.

Table ES-1. Water use per hydraulically fractured well between January 2011 and February 2013. Medians and percentiles were calculated from data submitted to FracFocus 1.0 (Appendix B).

State	Number of FracFocus 1.0 Disclosures	Median Volume per Well (gallons)	10th percentile (gallons)	90th percentile (gallons)
Arkansas	1,423	5,259,965	3,234,963	7,121,249
California	711	76,818	21,462	285,306
Colorado	4,898	463,462	147,353	3,092,024
Kansas	121	1,453,788	10,836	2,227,926
Louisiana	966	5,077,863	1,812,099	7,945,630
Montana	207	1,455,757	367,326	2,997,552
New Mexico	1,145	175,241	35,638	1,871,666
North Dakota	2,109	2,022,380	969,380	3,313,482
Ohio	146	3,887,499	2,885,568	5,571,027
Oklahoma	1,783	2,591,778	1,260,906	7,402,230
Pennsylvania	2,445	4,184,936	2,313,649	6,615,981
Texas	16,882	1,420,613	58,709	6,115,195
Utah	1,406	302,075	76,286	769,360
West Virginia	273	5,012,238	3,170,210	7,297,080
Wyoming	1,405	322,793	5,727	1,837,602

resources, whereas operations in the arid and semi-arid western United States generally rely on groundwater or surface water. Geographic differences in water use for hydraulic fracturing are illustrated in Figure ES-4, which shows that most of the water used for hydraulic fracturing in the Marcellus Shale region of the Susquehanna River Basin came from surface water resources between approximately 2008 and 2013. In comparison, less than half of the water used for hydraulic fracturing in the Barnett Shale region of Texas came from surface water resources between approximately 2011 and 2013.

Hydraulic fracturing wastewater and other lower-quality water can also be used in hydraulic fracturing fluids to offset the need for fresh water, although the proportion of injected fluid that is reused hydraulic fracturing wastewater varies by location (Figure ES-4).[1] Overall, the proportion of water used in hydraulic fracturing that comes from reused hydraulic fracturing wastewater appears to be low. In a survey of literature values from 10 states, basins, or plays, the median percentage of the injected fluid volume that came from reused hydraulic fracturing wastewater was 5% between approximately 2008 and 2014.[2] There was an increase in the reuse of hydraulic fracturing wastewater as a percentage of the injected hydraulic fracturing fluid in both Pennsylvania and West Virginia between approximately 2008 and 2014. This increase is likely due to the limited availability of Class II wells, which are commonly used to dispose of oil and gas wastewater, and the costs of trucking wastewater to Ohio, where Class II wells are

[1] Reused hydraulic fracturing wastewater as a percentage of injected fluid differs from the percentage of produced water that is managed through reuse in other hydraulic fracturing operations. For example, in the Marcellus Shale region of the Susquehanna River Basin, approximately 14% of injected fluid was reused hydraulic fracturing wastewater, while approximately 90% of produced water was managed through reuse in other hydraulic fracturing operations (Figure ES-4a).

[2] See Section 4.2 in Chapter 4.

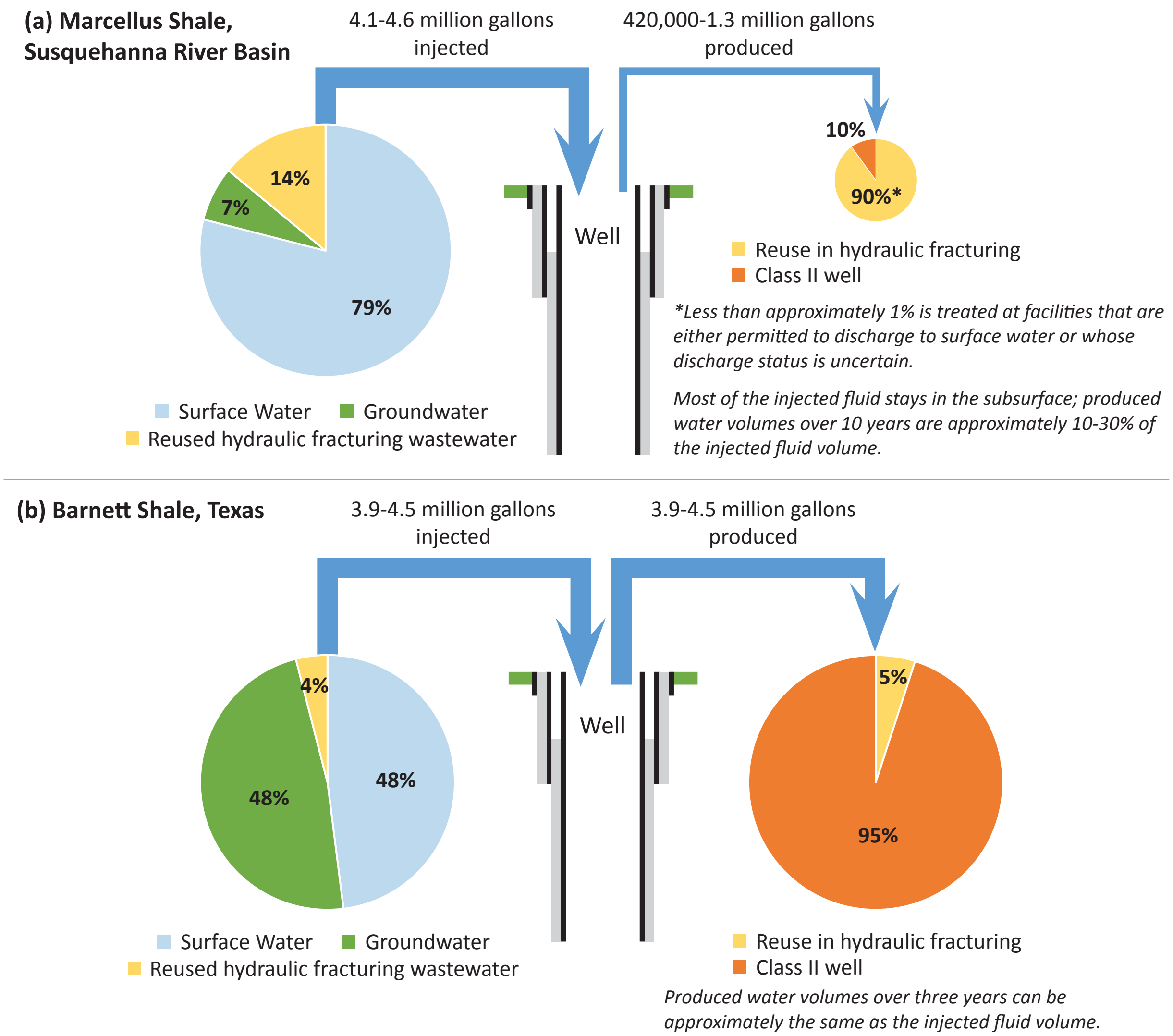


Figure ES-4. Water budgets illustrative of hydraulic fracturing water management practices in (a) the Marcellus Shale in the Susquehanna River Basin between approximately 2008 and 2013 and (b) the Barnett Shale in Texas between approximately 2011 and 2013. Class II wells are used to inject wastewater associated with oil and gas production underground and are regulated under the Underground Injection Control Program of the Safe Drinking Water Act. Data sources are described in Figure 10-1 in Chapter 10.

more prevalent.[1] Class II wells are also prevalent in Texas, and the reuse of wastewater in hydraulic fracturing fluids in the Barnett Shale appears to be lower than in the Marcellus Shale (Figure ES-4).

Because the same water resource can be used to support hydraulic fracturing and to provide drinking water, withdrawals for hydraulic fracturing can directly impact drinking water resources by changing the quantity or quality of the remaining water. Although every water withdrawal affects water quantity, we focused on water withdrawals that have the potential to significantly impact drinking water re-

[1] See Chapter 8 for additional information on Class II wells.

sources by limiting the availability of drinking water or altering its quality. Water withdrawals for a single hydraulically fractured oil and gas production well are not expected to significantly impact drinking water resources, because the volume of water needed to hydraulically fracture a single well is unlikely to limit the availability of drinking water or alter its quality. If, however, multiple oil and gas production wells are located within an area, the total volume of water needed to hydraulically fracture all of the wells has the potential to be a significant portion of the water available and impacts on drinking water resources can occur.

To assess whether hydraulic fracturing operations are a relatively large or small user of water, we compared water use for hydraulic fracturing to total water use at the county level (Text Box ES-5). In most counties studied, the average annual water volumes reported in FracFocus 1.0 were generally less than 1% of total water use. This suggests that hydraulic fracturing operations represented a relatively small user of water in most counties. There were exceptions, however. Average annual water volumes reported in FracFocus 1.0 were 10% or more of total water use in 26 of the 401 counties studied, 30% or more in nine counties, and 50% or more in four counties.[1] In these counties, hydraulic fracturing operations represented a relatively large user of water.

The above results suggest that hydraulic fracturing operations can significantly increase the volume of water withdrawn in particular areas. Increased water withdrawals can result in significant impacts on drinking water resources if there is insufficient water available in the area to accommodate all users. To assess the potential for these impacts, we compared hydraulic fracturing water use to estimates of water availability at the county level.[2] In most counties studied, average annual water volumes reported for hydraulic fracturing were less than 1% of the estimated annual volume of readily-available fresh water. However, average annual water volumes reported for hydraulic fracturing were greater than the estimated annual volume of readily-available fresh water in 17 counties in Texas. This analysis suggests that there was enough water available annually to support the level of hydraulic fracturing reported to FracFocus 1.0 in most, but not all, areas of the country. This observation does not preclude the possibility of local impacts in other areas of the country, nor does it indicate that local impacts have occurred or will occur in the 17 counties in Texas. To better understand whether local impacts have occurred, and the factors that affect those impacts, local-level studies, such as the ones described below, are needed.

Local impacts on drinking water quantity have occurred in areas with increased hydraulic fracturing activity. In 2011, for example, drinking water wells in an area overlying the Haynesville Shale ran out of water due to higher than normal groundwater withdrawals and drought (Louisiana Ground Water Resources Commission, 2012). Water withdrawals for hydraulic fracturing contributed to these conditions, along with other water users and the lack of precipitation. Groundwater impacts have also been reported in Texas. In a detailed case study, Scanlon et al. (2014) estimated that groundwater levels in approximately 6% of the area studied dropped by 100 feet (31 meters) to 200 feet (61 meters) or more after hydraulic fracturing activity increased in 2009.

In contrast, studies in the Upper Colorado and Susquehanna River basins found minimal impacts on drinking water resources from hydraulic fracturing. In the Upper Colorado River Basin, the EPA found that high-quality water produced from oil and gas wells in the Piceance tight sands provided nearly all of the water for hydraulic fracturing in the study area (U.S. EPA,

[1] Hydraulic fracturing water consumption estimates followed the same general pattern as the water use estimates presented here, but with slightly larger percentages in each category (Section 4.4 in Chapter 4).

[2] County-level water availability estimates were derived from the Tidwell et al. (2013) estimates of water availability for siting new thermoelectric power plants (see Text Box 4-2 in Chapter 4 for details). The county-level water availability estimates used in this report represent the portion of water available to new users within a county.

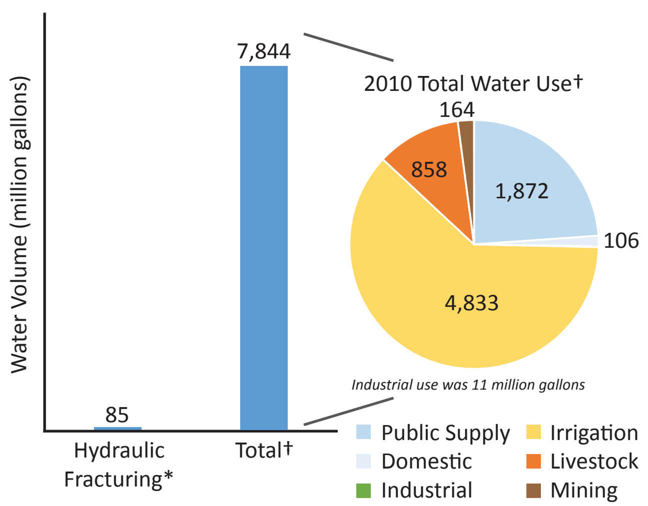
Text Box ES-5: County-Level Water Use for Hydraulic Fracturing

To assess whether hydraulic fracturing operations are a relatively large or small user of water, the average annual water use for hydraulic fracturing in 2011 and 2012 was compared, at the county-level, to total water use in 2010.

For most counties studied, average annual water volumes reported for individual counties in FracFocus 1.0 were less than 1% of total water use in those counties. But in some counties, hydraulic fracturing operations reported in FracFocus 1.0 represented a relatively large user of water.

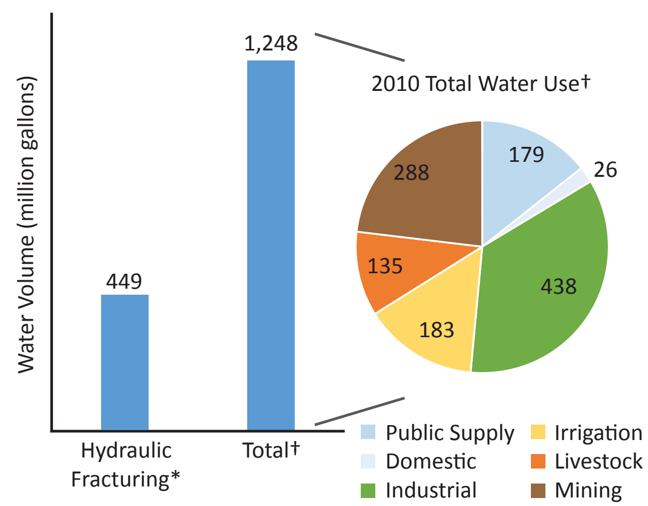
Examples of Water Use in Two Counties: Wilson County, Texas, and Mountrail County, North Dakota

Wilson County, Texas
44 wells reported in FracFocus 1.0



Depending on local water availability, hydraulic fracturing water withdrawals may be **less likely** to significantly impact drinking water resources under this kind of scenario.

Mountrail County, North Dakota
508 wells reported in FracFocus 1.0



Depending on local water availability, hydraulic fracturing water withdrawals may be **more likely** to significantly impact drinking water resources under this kind of scenario.

*Hydraulic fracturing water use is a function of the water use per well and the total number of wells hydraulically fractured within a county. Average annual water use for hydraulic fracturing was calculated at the county-level using data reported in FracFocus 1.0 in 2011 and 2012 (Appendix B).

†The U.S. Geological Survey compiles national water use estimates every five years in the National Water Census. Total water use at the county-level was obtained from the most recent census, which was conducted in 2010 (Maupin et al., 2014).

2010 Total Water Use Categories

Public supply	Water withdrawn by public and private water suppliers that provide water to at least 25 people or have a minimum of 15 connections
Domestic	Self-supplied water withdrawals for indoor household purposes such as drinking, food preparation, bathing, washing clothes and dishes, flushing toilets, and outdoor purposes such as watering lawns and gardens
Industrial	Water used for fabrication, processing, washing, and cooling
Irrigation	Water that is applied by an irrigation system to assist crop and pasture growth or to maintain vegetation on recreational lands (e.g., parks and golf courses)
Livestock	Water used for livestock watering, feedlots, dairy operations, and other on-farm needs
Mining	Water used for the extraction of naturally-occurring minerals, including solids (e.g., coal, sand, gravel, and other ores), liquids (e.g., crude petroleum), and gases (e.g., natural gas)

2015b). Due to this high reuse rate, the EPA did not identify any locations in the study area where hydraulic fracturing contributed to locally high water use. In the Susquehanna River Basin, multiple studies and state reports have identified the potential for hydraulic fracturing water withdrawals in the Marcellus Shale to impact surface water resources. Evidence suggests, however, that current water management strategies, including passby flows and reuse of hydraulic fracturing wastewater, help protect streams from depletion by hydraulic fracturing water withdrawals. A passby flow is a prescribed, low-streamflow threshold below which water withdrawals are not allowed.

The above examples highlight factors that can affect the frequency or severity of impacts on drinking water resources from hydraulic fracturing water withdrawals. In particular, areas of the United States that rely on declining groundwater resources are vulnerable to more frequent and more severe impacts from all water withdrawals, including withdrawals for hydraulic fracturing. Extensive groundwater withdrawals can limit the availability of belowground drinking water resources and can also change the quality of the water remaining in the resource. Because groundwater recharge rates can be low, impacts can last for many years. Seasonal or long-term drought can also make impacts more frequent and more severe for groundwater and surface water resources. Hot, dry weather reduces or prevents groundwater recharge and depletes surface water bodies, while water demand often increases simultaneously (e.g., for irrigation). This combination of factors—high hydraulic fracturing water use and relatively low water availability due to declining groundwater resources and/or frequent drought—was found to be present in southern and western Texas.

Water management strategies can also affect the frequency and severity of impacts on drinking water resources from hydraulic fracturing water withdrawals. These strategies include using hydraulic fracturing wastewater or brackish groundwater for hydraulic fracturing, transitioning from limited groundwater resources to more abundant surface water resources, and using passby flows to control water withdrawals from surface water resources. Examples of these water management strategies can be found throughout the United States. In western and southern Texas, for example, the use of brackish water is currently reducing impacts on fresh water sources, and could, if increased, reduce future impacts. Louisiana and North Dakota have encouraged well operators to withdraw water from surface water resources instead of high-quality groundwater resources. And, as described above, the Susquehanna River Basin Commission limits surface water withdrawals during periods of low stream flow.

Water Acquisition Conclusions

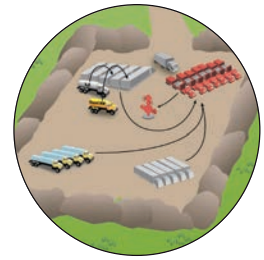
With notable exceptions, hydraulic fracturing uses a relatively small percentage of water when compared to total water use and availability at large geographic scales. Despite this, hydraulic fracturing water withdrawals can affect the quantity and quality of drinking water resources by changing the balance between the demand on local water resources and the availability of those resources. Changes that have the potential to limit the availability of drinking water or alter its quality are more likely to occur in areas with relatively high hydraulic fracturing water withdrawals and low water availability, particularly due to limited or declining groundwater resources. Water management strategies (e.g., encouragement of alternative water sources or water withdrawal restrictions) can reduce the frequency or severity of impacts on drinking water resources from hydraulic fracturing water withdrawals.

Chemical Mixing

The mixing of a base fluid, proppant, and additives at the well site to create hydraulic fracturing fluids.

<ins>**Relationship to Drinking Water Resources**</ins>

Spills of additives and hydraulic fracturing fluids can reach groundwater and surface water resources.



Hydraulic fracturing fluids are engineered to create and grow fractures in the targeted rock formation and to carry proppant through the oil and gas production well into the newly-created fractures. Hydraulic fracturing fluids are typically made up of base fluids, proppant, and additives. Base fluids make up the largest proportion of hydraulic fracturing fluids by volume. As illustrated in Text Box ES-6, base fluids can be a single substance (e.g., water in the slickwater example) or can be a mixture of substances (e.g., water and nitrogen in the energized fluid example). The EPA's analysis of hydraulic fracturing fluid data reported to FracFocus 1.0 suggests that water was the most commonly used base fluid between January 2011 and February 2013 (U.S. EPA, 2015a). Non-water substances, such as gases and hydrocarbon liquids, were reported to be used alone or blended with water to form a base fluid in fewer than 3% of wells in FracFocus 1.0.

Proppant makes up the second largest proportion of hydraulic fracturing fluids (Text Box ES-6). Sand (i.e., quartz) was the most commonly reported proppant between January 2011 and February 2013, with 98% of wells in FracFocus 1.0 reporting sand as the proppant (U.S. EPA, 2015a). Other proppants can include man-made or specially engineered particles, such as high-strength ceramic materials or sintered bauxite.[1]

Additives generally make up the smallest proportion of the overall composition of hydraulic fracturing fluids (Text Box ES-6), yet have the greatest potential to impact the quality of drinking water resources compared to proppant and base fluids. Additives, which can be a single chemical or a mixture of chemicals, are added to the base fluid to change its properties (e.g., adjust pH, increase fluid thickness, or limit bacterial growth). The choice of which additives to use depends on the characteristics of the targeted rock formation (e.g., rock type, temperature, and pressure), the economics and availability of desired additives, and well operator or service company preferences and experience.

The variability of additives, both in their purpose and chemical composition, suggests that a large number of different chemicals may be used in hydraulic fracturing fluids across the United States. The EPA identified 1,084 chemicals that were reported to have been used in hydraulic fracturing fluids between 2005 and 2013.[2,3] The EPA's analysis of FracFocus 1.0 data indicates that between 4 and 28 chemicals were used per well between January 2011 and February 2013 and that no single chemical was used in all wells (U.S. EPA, 2015a). Three chemicals—methanol, hydrotreated light petroleum distillates, and hydro-

[1] Sintered bauxite is crushed and powdered bauxite that is fused into spherical beads at high temperatures.

[2] This list includes 1,084 unique Chemical Abstracts Service Registration Numbers (CASRNs), which can be assigned to a single chemical (e.g., hydrochloric acid) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates). Throughout this report, we refer to the substances identified by unique CASRNs as "chemicals."

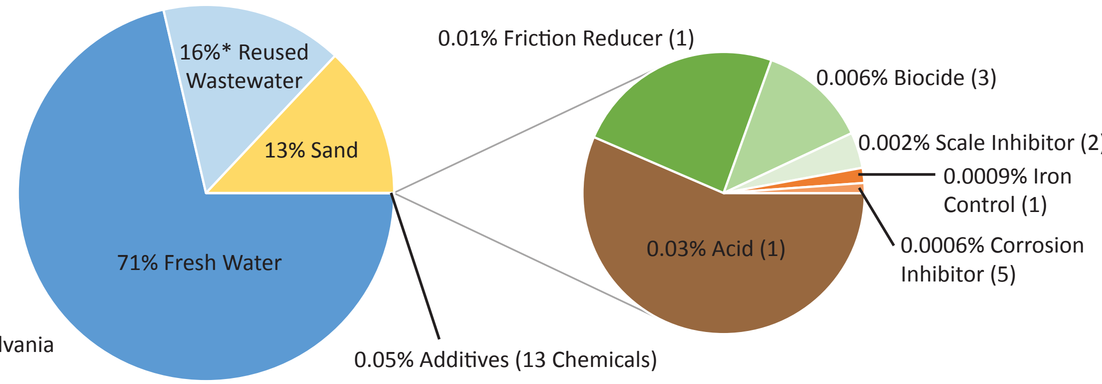
[3] Dayalu and Konschnik (2016) identified 995 unique CASRNs from data submitted to FracFocus between March 9, 2011, and April 13, 2015. Two hundred sixty-three of these CASRNs are not on the list of unique CASRNs identified by the EPA (Appendix H). Only one of the 263 chemicals was reported at greater than 1% of wells, which suggests that these chemicals were used at only a few sites.

Text Box ES-6: Examples of Hydraulic Fracturing Fluids

Hydraulic fracturing fluids are engineered to create and extend fractures in the targeted rock formation and to carry proppant through the production well into the newly-created fractures. While there is no universal hydraulic fracturing fluid, there are general types of hydraulic fracturing fluids. Two types of hydraulic fracturing fluids are described below.

Slickwater

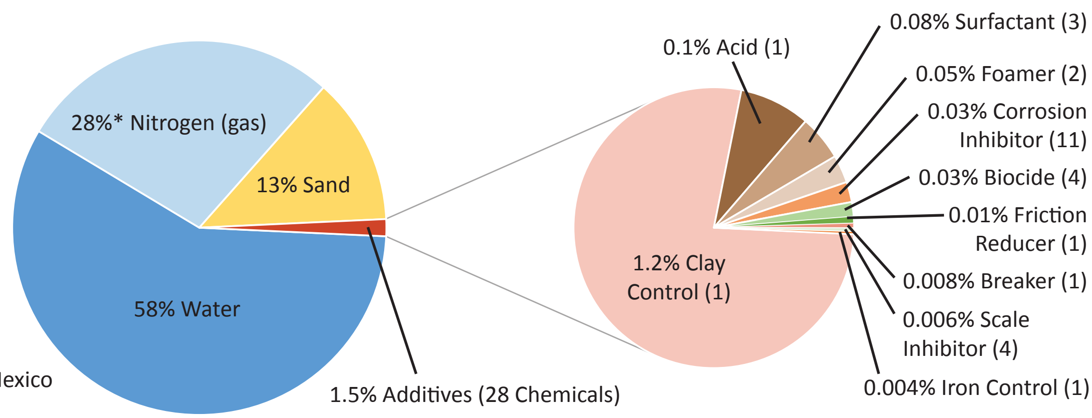
Slickwater hydraulic fracturing fluids are water-based fluids that generally contain a friction reducer. The friction reducer makes it easier for the fluid to be pumped down the oil and gas production well at high rates. Slickwater is commonly used to hydraulically fracture shale formations.



Bradford County, Pennsylvania
Well depth = 7,255 feet
Total water volume = 4,763,000 gallons

Energized Fluid

Energized fluids are mixtures of liquids and gases. They can be used for hydraulic fracturing in under-pressured gas formations.



Rio Arriba County, New Mexico
Well depth = 7,640 feet
Total water volume = 105,000 gallons

**Maximum percent by mass of the total hydraulic fracturing fluid. Data obtained from FracFocus.org.*

Additive Dictionary

Acid	Dissolves minerals and creates pre-fractures in the rock
Biocide	Controls or eliminates bacteria in the hydraulic fracturing fluid
Breaker	Reduces the thickness of the hydraulic fracturing fluid
Clay control	Prevents swelling and migration of formation clays
Corrosion inhibitor	Protects iron and steel equipment from rusting
Foamer	Creates a foam hydraulic fracturing fluid
Friction reducer	Reduces friction between the hydraulic fracturing fluid and pipes during pumping
Iron control	Prevents the precipitation of iron-containing chemicals
Scale inhibitor	Prevents the formation of scale buildup within the well
Surfactant	Reduces the surface tension of the hydraulic fracturing fluid

Table ES-2. Chemicals reported in 10% or more of disclosures in FracFocus 1.0. Disclosures provided information on chemicals used at individual well sites between January 1, 2011, and February 28, 2013.

Chemical Name (CASRN)[a]	Percent of FracFocus 1.0 Disclosures[b]
Methanol (67-56-1)	72
Hydrotreated light petroleum distillates (64742-47-8)	65
Hydrochloric acid (7647-01-0)	65
Water (7732-18-5)[c]	48
Isopropanol (67-63-0)	47
Ethylene glycol (107-21-1)	46
Peroxydisulfuric acid, diammonium salt (7727-54-0)	44
Sodium hydroxide (1310-73-2)	39
Guar gum (9000-30-0)	37
Quartz (14808-60-7)[c]	36
Glutaraldehyde (111-30-8)	34
Propargyl alcohol (107-19-7)	33
Potassium hydroxide (1310-58-3)	29
Ethanol (64-17-5)	29
Acetic acid (64-19-7)	24
Citric acid (77-92-9)	24
2-Butoxyethanol (111-76-2)	21
Sodium chloride (7647-14-5)	21
Solvent naphtha, petroleum, heavy aromatic (64742-94-5)	21

Chemical Name (CASRN)[a]	Percent of FracFocus 1.0 Disclosures[b]
Naphthalene (91-20-3)	19
2,2-Dibromo-3-nitrilopropionamide (10222-01-2)	16
Phenolic resin (9003-35-4)	14
Choline chloride (67-48-1)	14
Methenamine (100-97-0)	14
Carbonic acid, dipotassium salt (584-08-7)	13
1,2,4-Trimethylbenzene (95-63-6)	13
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (68424-85-1)	12
Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture) (127087-87-0)	12
Formic acid (64-18-6)	12
Sodium chlorite (7758-19-2)	11
Nonyl phenol ethoxylate (9016-45-9)	11
Tetrakis(hydroxymethyl)phosphonium sulfate (55566-30-8)	11
Polyethylene glycol (25322-68-3)	11
Ammonium chloride (12125-02-9)	10
Sodium persulfate (7775-27-1)	10

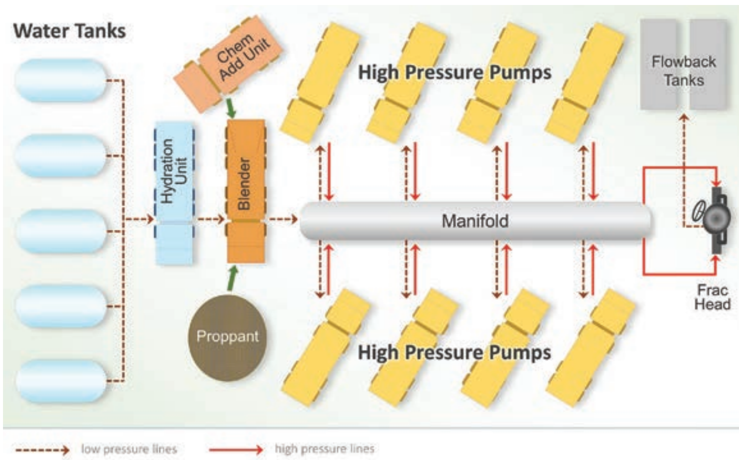
[a]"Chemical" refers to chemical substances with a single CASRN; these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates).
[b]Analysis considered 34,675 disclosures that met selected quality assurance criteria. See Table 5-2 in Chapter 5.
[c]Quartz and water were reported as ingredients in additives, in addition to proppants and base fluids.

chloric acid—were reported in 65% or more of the wells in FracFocus 1.0; 35 chemicals were reported in at least 10% of the wells (Table ES-2).

Concentrated additives are delivered to the well site and stored until they are mixed with the base fluid and proppant and pumped down the oil and gas production well (Text Box ES-7). While the overall concentration of additives in hydraulic fracturing fluids is generally small (typically 2% or less of the total volume of the injected fluid), the total volume of additives delivered to the well site can be large. Because over 1 million gallons (3.8 million liters) of hydraulic fracturing fluid are generally injected per well, thousands of gallons of additives can be stored on site and used during hydraulic fracturing.

As illustrated in Text Box ES-7, additives are often stored in multiple, closed containers [typically 200 gallons (760 liters) to 375 gallons (1,420 liters) per container] and moved around the site in hoses and tubing. This equipment is designed to contain additives and blended hydraulic fracturing fluid, but spills can occur. Changes in drinking water quality can occur if spilled fluids reach groundwater or surface water resources.

Text Box ES-7: Chemical Mixing Equipment



Source: Adapted from Olson (2011) and BJ Services Company (2009)

Typical Layout of Chemical Mixing Equipment

This illustration shows how the different pieces of equipment fit together to contain, mix, and inject hydraulic fracturing fluid into a production well.

Water, proppant, and additives are blended together and pumped to the manifold, where high pressure pumps transfer the fluid to the frac head.

Additives and proppant can be blended with water at different times and in different amounts during hydraulic fracturing. Thus, the composition of hydraulic fracturing fluids can vary during the hydraulic fracturing job.

Well Pad During Hydraulic Fracturing

Equipment set up for hydraulic fracturing.

Source: Schlumberger

Chemical Mixing Equipment Dictionary

Blender	Blends water, proppant, and additives
Chemical additive unit	Transports additives to the site and stores additives onsite
Flowback tanks	Stores liquid that returns to the surface after hydraulic fracturing
Frac head	Connects hydraulic fracturing equipment to the production well
High pressure pumps	Pressurize mixed fluids before injection into the production well
Hydration unit	Creates and stores gels used in some hydraulic fracturing fluids
Manifold	Transfers fluids from the blender to the frac head
Proppant	Stores proppant (often sand)
Water tanks	Stores water

Several studies have documented spills of hydraulic fracturing fluids or additives. Nearly all of these studies identified spills from state-managed spill databases. Data gathered for these studies suggest that spills of hydraulic fracturing fluids or additives were primarily caused by equipment failure or human error. For example, an EPA analysis of spill reports from nine state agencies, nine oil and gas well operators, and nine hydraulic fracturing service companies characterized 151 spills of hydraulic fracturing fluids or additives on or near well sites in 11 states between January 2006 and April 2012 (U.S. EPA, 2015c). These spills were primarily caused by equipment failure (34% of the spills) or human error (25%), and more than 30% of the spills were from fluid storage units (e.g., tanks, totes, and trailers). Similarly, a study of spills reported to the Colorado Oil and Gas Conservation Commission identified 125 spills during well stimulation (i.e., a part of the life of an oil and gas well that often, but not always, includes hydraulic fracturing) between January 2010 and August 2013 (COGCC, 2014). Of these spills, 51% were caused by human error and 46% were due to equipment failure.

Studies of spills of hydraulic fracturing fluids or additives provide insights on spill volumes, but little information on chemical-specific spill composition. Among the 151 spills characterized by the EPA, the median volume of fluid spilled was 420 gallons (1,600 liters), although the volumes spilled ranged from 5 gallons (19 liters) to 19,320 gallons (73,130 liters). Spilled fluids were often described as acids, biocides, friction reducers, crosslinkers, gels, and blended hydraulic fracturing fluid, but few specific chemicals were mentioned.[1] Considine et al. (2012) identified spills related to oil and gas development in the Marcellus Shale that occurred between January 2008 and August 2011 from Notices of Violations issued by the Pennsylvania Department of Environmental Protection. The authors identified spills greater than 400 gallons (1,500 liters) and spills less than 400 gallons (1,500 liters).

Spills of hydraulic fracturing fluids or additives have reached, and therefore impacted, surface water resources. Thirteen of the 151 spills characterized by the EPA were reported to have reached a surface water body (often creeks or streams). Among the 13 spills, reported spill volumes ranged from 28 gallons (105 liters) to 7,350 gallons (27,800 liters). Additionally, Brantley et al. (2014) and Considine et al. (2012) identified fewer than 10 total instances of spills of additives and/or hydraulic fracturing fluids greater than 400 gallons (1,500 liters) that reached surface waters in Pennsylvania between January 2008 and June 2013. Reported spill volumes for these spills ranged from 3,400 gallons (13,000 liters) to 227,000 gallons (859,000 liters).

Although impacts on surface water resources have been documented, site-specific studies that could be used to describe factors that affect the frequency or severity of impacts were not available. In the absence of such studies, we relied on fundamental scientific principles to identify factors that affect how hydraulic fracturing fluids and chemicals can move through the environment to drinking water resources. Because these factors influence whether spilled fluids reach groundwater and surface water resources, they affect the frequency and severity of impacts on drinking water resources from spills during the chemical mixing stage of the hydraulic fracturing water cycle.

The potential for spilled fluids to impact groundwater or surface water resources depends on the characteristics of the spill, the environmental fate and transport of the spilled fluid, and spill response activities (Figure ES-5). Site-specific characteristics affect how spilled liquids move through soil into the subsurface or over the land surface. Generally, highly permeable soils or fractured rock can allow spilled liquids to move quickly into and through the subsurface, limiting the opportunity for spilled liquids to move over land to surface water resources. In low permeability soils, spilled liquids are less able to move into the subsurface and are more likely to move over the

[1] A crosslinker is an additive that increases the thickness of gelled fluids by connecting polymer molecules in the gelled fluid.

land surface. In either case, the volume spilled and the distance between the location of the spill and nearby water resources affects whether spilled liquids reach drinking water resources. Large-volume spills are generally more likely to reach drinking water resources because they are more likely to be able to travel the distance between the location of the spill and nearby water resources.

In general, chemical and physical properties, which depend on the identity and structure of a chemical, control whether spilled chemicals evaporate, stick to soil particles, or move with water. The EPA identified measured or estimated chemical and physical properties for 455 of the 1,084 chemicals used in hydraulic fracturing fluids between 2005 and 2013.[1] The properties of these chemicals varied

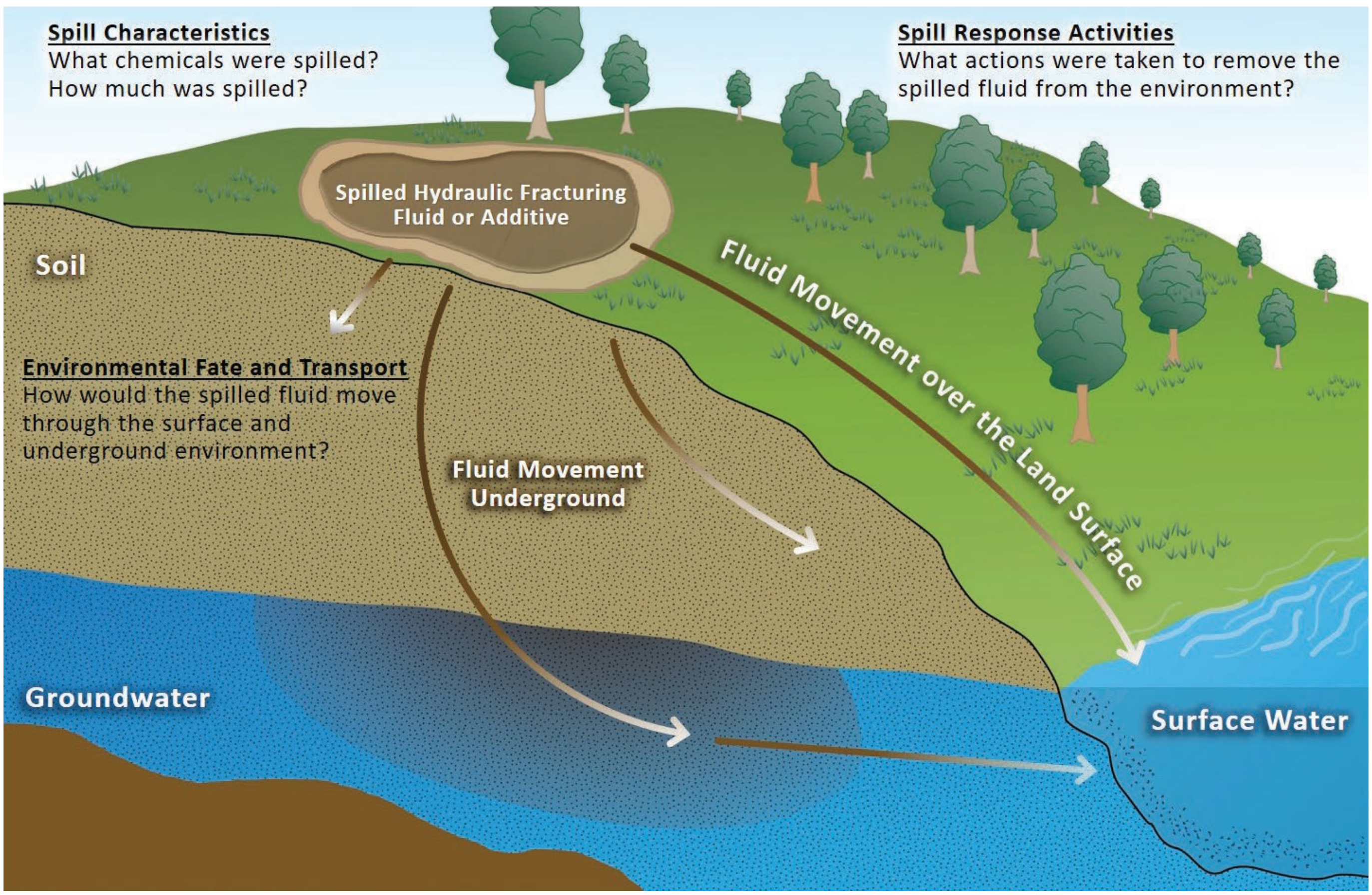


Figure ES-5. Generalized depiction of factors that influence whether spilled hydraulic fracturing fluids or additives reach drinking water resources, including spill characteristics, environmental fate and transport, and spill response activities.

[1] Chemical and physical properties were identified using EPI Suite™. EPI Suite™ is a collection of chemical and physical property and environmental fate estimation programs developed by the EPA and Syracuse Research Corporation. It can be used to estimate chemical and physical properties of individual organic compounds. Of the 1,084 hydraulic fracturing fluid chemicals identified by the EPA, 629 were not individual organic compounds, and thus EPI Suite™ could not be used to estimate their chemical and physical properties.

widely, from chemicals that are more likely to move quickly through the environment with a spilled liquid to chemicals that are more likely to move slowly through the environment because they stick to soil particles.[1] Chemicals that move slowly through the environment may act as longer-term sources of contamination if spilled.

Spill prevention practices and spill response activities are designed to prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids. Spill prevention and response activities are influenced by federal, state, and local regulations and company practices. Spill prevention practices include secondary containment systems (e.g., liners and berms), which are designed to contain spilled fluids and prevent them from reaching soil, groundwater, or surface water. Spill response activities include activities taken to stop the spill, contain spilled fluids (e.g., the deployment of emergency containment systems), and clean up spilled fluids (e.g., removal of contaminated soil). It was beyond the scope of this report to evaluate the implementation and efficacy of spill prevention practices and spill response activities.

The severity of impacts on water quality from spills of hydraulic fracturing fluids or additives depends on the identity and amount of chemicals that reach groundwater or surface water resources, the toxicity of the chemicals, and the characteristics of the receiving water resource.[2] Characteristics of the receiving groundwater or surface water resource (e.g., water resource size and flow rate) can affect the magnitude and duration of impacts by reducing the concentration of spilled chemicals in a drinking water resource. Impacts on groundwater resources have the potential to be more severe than impacts on surface water resources because it takes longer to naturally reduce the concentration of chemicals in groundwater and because it is generally difficult to remove chemicals from groundwater resources. Due to a lack of data, particularly in terms of groundwater monitoring after spill events, little is publicly known about the severity of drinking water impacts from spills of hydraulic fracturing fluids or additives.

Chemical Mixing Conclusions

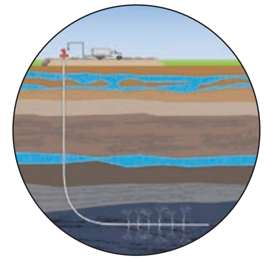
Spills of hydraulic fracturing fluids and additives during the chemical mixing stage of the hydraulic fracturing water cycle have reached surface water resources in some cases and have the potential to reach groundwater resources. Although the available data indicate that spills of various volumes can reach surface water resources, large volume spills are more likely to travel longer distances to nearby groundwater or surface water resources. Consequently, large volume spills likely increase the frequency of impacts on drinking water resources. Large volume spills, particularly of concentrated additives, are also likely to result in more severe impacts on drinking water resources than small volume spills because they can deliver a large quantity of potentially hazardous chemicals to groundwater or surface water resources. Impacts on groundwater resources are likely to be more severe than impacts on surface water resources because of the inherent characteristics of groundwater. Spill prevention and response activities are designed to prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids.

[1] These results describe how some hydraulic fracturing chemicals behave in infinitely dilute aqueous solutions, which is a simplified approximation of the real-world mixtures found in hydraulic fracturing fluids. The presence of other chemicals in a mixture can affect the fate and transport of a chemical.

[2] Human health hazards associated with hydraulic fracturing fluid chemicals are discussed in Chapter 9 and summarized in the "Chemicals in the Hydraulic Fracturing Water Cycle" section below.

Well Injection

The injection and movement of hydraulic fracturing fluids through the oil and gas production well and in the targeted rock formation.



Relationship to Drinking Water Resources

Belowground pathways, including the production well itself and newly-created fractures, can allow hydraulic fracturing fluids or other fluids to reach underground drinking water resources.

Hydraulic fracturing fluids primarily move along two pathways during the well injection stage: the oil and gas production well and the newly-created fracture network. Oil and gas production wells are designed and constructed to move fluids to and from the targeted rock formation without leaking and to prevent fluid movement along the outside of the well. This is generally accomplished by installing multiple layers of casing and cement within the drilled hole (Text Box ES-2), particularly where the well intersects oil-, gas-, and/or water-bearing rock formations. Casing and cement, in addition to other well components (e.g., packers), can control hydraulic fracturing fluid movement by creating a preferred flow pathway (i.e., inside the casing) and preventing unintentional fluid movement (e.g., from the inside of the casing to the surrounding environment or vertically along the well from the targeted rock formation to shallower formations).[1] An EPA survey of oil and gas production wells hydraulically fractured between approximately September 2009 and September 2010 suggests that hydraulically fractured wells are often, but not always, constructed with multiple casings that have varying amounts of cement surrounding each casing (U.S. EPA, 2015d). Among the wells surveyed, the most common number of casings per well was two: surface casing and production casing (Text Box ES-2). The presence of multiple cemented casings that extend from the ground surface to below the designated drinking water resource is one of the primary well construction features that protects underground drinking water resources.

During hydraulic fracturing, a well is subjected to greater pressure and temperature changes than during any other activity in the life of the well. As hydraulic fracturing fluid is injected into the well, the pressure applied to the well increases until the targeted rock formation fractures; then pressure decreases. Maximum pressures applied to wells during hydraulic fracturing have been reported to range from less than 2,000 pounds per square inch (psi) [14 megapascals (MPa)] to approximately 12,000 psi (83 MPa).[2] A well can also experience temperature changes as cooler hydraulic fracturing fluid enters the warmer well. In some cases, casing temperatures have been observed to drop from 212°F (100°C) to 64°F (18°C). A well can experience multiple pressure and temperature cycles if hydraulic fracturing is done in multiple stages or if a well is re-fractured.[3] Casing, cement, and other well components need to be able to withstand these changes in pressure and temperature, so that hydraulic fracturing fluids can flow to the targeted rock formation without leaking.

The fracture network created during hydraulic fracturing is the other primary pathway along

[1] Packers are mechanical devices installed with casing. Once the casing is set in the drilled hole, packers swell to fill the space between the outside of the casing and the surrounding rock or casing.

[2] For comparison, average atmospheric pressure is approximately 15 psi.

[3] In a multi-stage hydraulic fracturing operation, specific parts of the well are isolated and hydraulically fractured until the total desired length of the well has been hydraulically fractured.

which hydraulic fracturing fluids move. Fracture growth during hydraulic fracturing is complex and depends on the characteristics of the targeted rock formation and the characteristics of the hydraulic fracturing operation. In general, rock characteristics, particularly the natural stresses placed on the targeted rock formation due to the weight of the rock above, affect how the rock fractures, including whether newly-created fractures grow vertically (i.e., perpendicular to the ground surface) or horizontally (i.e., parallel to the ground surface) (Text Box ES-8). Because hydraulic fracturing fluids are used to create and grow fractures, fracture growth during hydraulic fracturing can be controlled by limiting the rate and volume of hydraulic fracturing fluid injected into the well.

Publicly available data on fracture growth are currently limited to microseismic and tiltmeter data collected during hydraulic fracturing operations in five shale plays in the United States. Analyses of these data by Fisher and Warpinski (2012) and Davies et al. (2012) indicate that the direction of fracture growth generally varied with depth and that upward vertical fracture growth was often on the order of tens to hundreds of feet in the shale formations studied (Text Box ES-8). One percent of the fractures had a fracture height greater than 1,148 feet (350 meters), and the maximum fracture height among all of the data reported was 1,929 feet (588 meters). These reported fracture heights suggest that some fractures can grow out of the targeted rock formation and into an overlying formation. It is unknown whether these observations apply to other hydraulically fractured rock formations because similar data from hydraulic fracturing operations in other rock formations are not currently available to the public.

The potential for hydraulic fracturing fluids to reach, and therefore impact, underground drinking water resources is related to the pathways along which hydraulic fracturing fluids primarily move during hydraulic fracturing: the oil and gas production well itself and the fracture network created during hydraulic fracturing. Because the well can be a pathway for fluid movement, the mechanical integrity of the well is an important factor that affects the frequency and severity of impacts from the well injection stage of the hydraulic fracturing water cycle.[1] A well with insufficient mechanical integrity can allow unintended fluid movement, either from the inside to the outside of the well (pathway 1 in Figure ES-6) or vertically along the outside of the well (pathways 2-5). The existence of one or more of these pathways can result in impacts on drinking water resources if hydraulic fracturing fluids reach groundwater resources. Impacts on drinking water resources can also occur if gases or liquids released from the targeted rock formation or other formations during hydraulic fracturing travel along these pathways to groundwater resources.

The pathways shown in Figure ES-6 can exist because of inadequate well design or construction (e.g., incomplete cement around the casing where the well intersects with water-, oil-, or gas-bearing formations) or can develop over the well's lifetime, including during hydraulic fracturing. In particular, casing and cement can degrade over the life of the well because of exposure to corrosive chemicals, formation stresses, and operational stresses (e.g., pressure and temperature changes during hydraulic fracturing). As a result, some hydraulically fractured oil and gas production wells may develop one or more of the pathways shown in Figure ES-6. Changes in mechanical integrity over time have implications for older wells that are hydraulically fractured because these wells may not be able to withstand the stresses applied during hydraulic fracturing. Older wells may also be hydraulically fractured at shallower depths, where cement around the casing may be inadequate or missing.

Examples of mechanical integrity problems have been documented in hydraulically fractured oil and gas production wells. In one case, hydraulic

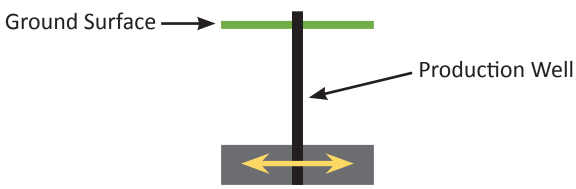
[1] Mechanical integrity is the absence of significant leakage within or outside of the well components.

Text Box ES-8: Fracture Growth

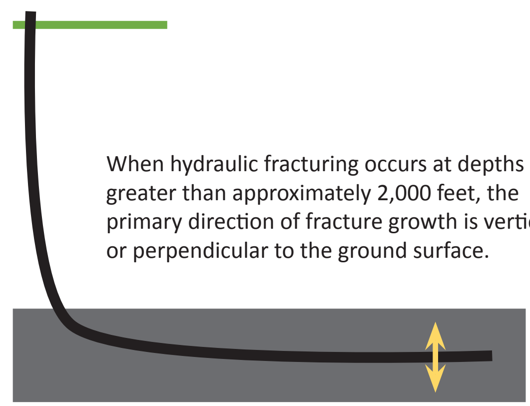
Fracture growth during hydraulic fracturing is complex and depends on the characteristics of the targeted rock formation and the characteristics of the hydraulic fracturing operation.

Primary Direction of Fracture Growth

In general, the weight of the rock above the point of hydraulic fracturing affects the primary direction of fracture growth. Therefore, the depth at which hydraulic fracturing occurs affects whether fractures grow vertically or horizontally.



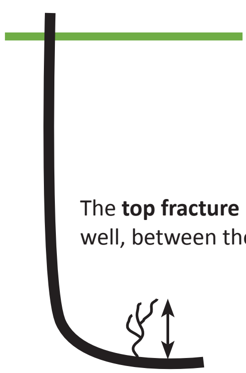
When hydraulic fracturing occurs at depths less than approximately 2,000 feet, the primary direction of fracture growth is horizontal, or parallel to the ground surface.



When hydraulic fracturing occurs at depths greater than approximately 2,000 feet, the primary direction of fracture growth is vertical, or perpendicular to the ground surface.

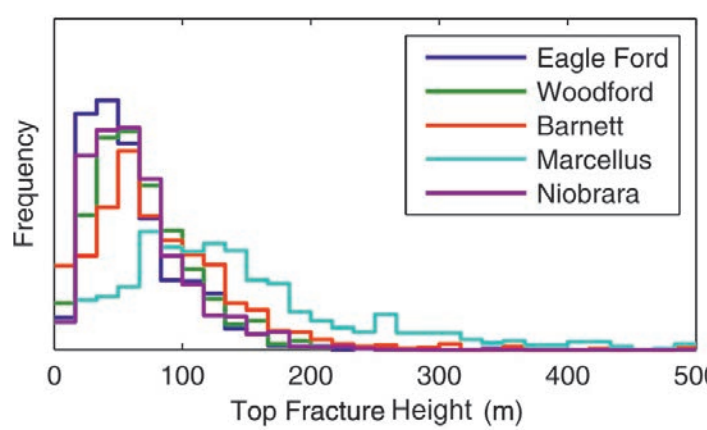
Fracture Height

Fisher and Warpinski (2012) and Davies et al. (2012) analyzed microseismic and tiltmeter data collected during thousands of hydraulic fracturing operations in the Barnett, Eagle Ford, Marcellus, Niobrara, and Woodford shale plays. Their data provide information on fracture heights in shale. Top fracture heights varied between shale plays and within individual shale plays.



The **top fracture height** is the vertical distance upward from the well, between the fracture tip and the well.

Shale Play	Approximate Median Top Fracture Height [feet (meters)]
Eagle Ford	130 (40)
Woodford	160 (50)
Barnett	200 (60)
Marcellus	400 (120)
Niobrara	160 (50)



Source: Davies et al. (2012)

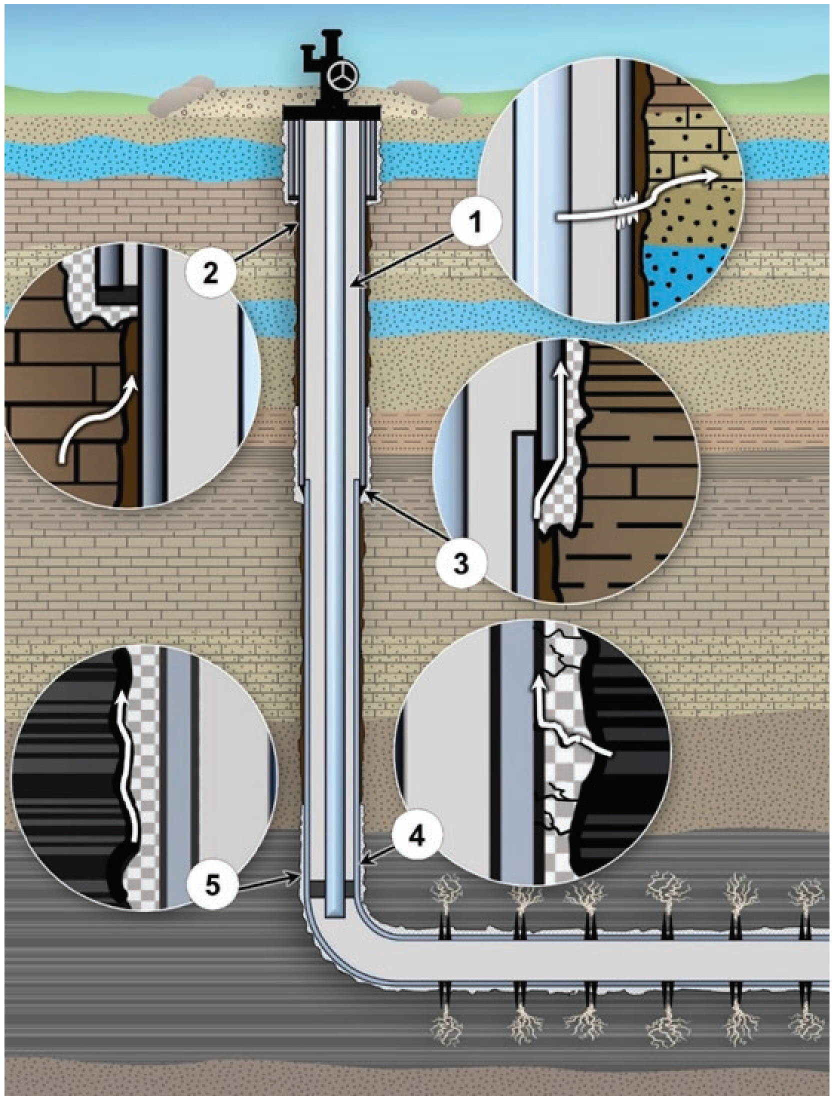


Figure ES-6. Potential pathways for fluid movement in a cemented well. These pathways (represented by the white arrows) include: (1) a casing and tubing leak into the surrounding rock, (2) an uncemented annulus (i.e., the space behind the casing), (3) microannuli between the casing and cement, (4) gaps in cement due to poor cement quality, and (5) microannuli between the cement and the surrounding rock. This figure is intended to provide a conceptual illustration of pathways that can be present in a well and is not to scale.

fracturing of an inadequately cemented gas well in Bainbridge Township, Ohio, contributed to the movement of methane into local drinking water resources.[1] In another case, an inner string of casing burst during hydraulic fracturing of an oil well near Killdeer, North Dakota, resulting in a release of hydraulic fracturing fluids and formation fluids that impacted a groundwater resource.

The potential for hydraulic fracturing fluids or other fluids to reach underground drinking water resources is also related to the fracture network created during hydraulic fracturing. Because fluids

[1] Although ingestion of methane is not considered to be toxic, methane can pose a physical hazard. Methane can accumulate to explosive levels when allowed to exsolve (degas) from groundwater in closed environments.

travel through the newly-created fractures, the location of these fractures relative to underground drinking water resources is an important factor affecting the frequency and severity of potential impacts on drinking water resources. Data on the relative location of induced fractures to underground drinking water resources are generally not available, because fracture networks are infrequently mapped and because there can be uncertainty in the depth of the bottom of the underground drinking water resource at a specific location.

Without these data, we were often unable to determine with certainty whether fractures created during hydraulic fracturing have reached underground drinking water resources. Instead, we considered the vertical separation distance between hydraulically fractured rock formations and the bottom of underground drinking water resources. Based on computer modeling studies, Birdsell et al. (2015) concluded that it is less likely that hydraulic fracturing fluids would reach an overlying drinking water resource if (1) the vertical separation distance between the targeted rock formation and the drinking water resource is large and (2) there are no open pathways (e.g., natural faults or fractures, or leaky wells). As the vertical separation distance between the targeted rock formation and the underground drinking water resource decreases, the likelihood of upward migration of hydraulic fracturing fluids to the drinking water resource increases (Birdsell et al., 2015).

Figure ES-7 illustrates how the vertical separation distance between the targeted rock formation and underground drinking water resources can vary across the United States. The two example environments depicted in panels a and b represent the range of separation distances shown in panel c. In Figure ES-7a, there are thousands of feet between the bottom of the underground drinking water resource and the hydraulically fractured rock formation. These conditions are generally reflective of deep shale formations (e.g., Haynesville Shale), where oil and gas production wells are first drilled vertically and then horizontally along the targeted rock formation. Microseismic data and modeling studies suggest that, under these conditions, fractures created during hydraulic fracturing are unlikely to grow through thousands of feet of rock into underground drinking water resources.

When drinking water resources are co-located with oil and gas resources and there is no vertical separation between the hydraulically fractured rock formation and the bottom of the underground drinking water resource (Figure ES-7b), the injection of hydraulic fracturing fluids impacts the quality of the drinking water resource. According to the information examined in this report, the overall occurrence of hydraulic fracturing within a drinking water resource appears to be low, with the activity generally concentrated in some areas in the western United States (e.g., the Wind River Basin near Pavillion, Wyoming, and the Powder River Basin of Montana and Wyoming).[1] Hydraulic fracturing within drinking water resources introduces hydraulic fracturing fluid into formations that may currently serve, or in the future could serve, as a drinking water source for public or private use. This is of concern in the short-term if people are currently using these formations as a drinking water supply. It is also of concern in the long-term, because drought or other conditions may necessitate the future use of these formations for drinking water.

Regardless of the vertical separation between the targeted rock formation and the underground drinking water resource, the presence of other wells near hydraulic fracturing operations can increase the potential for hydraulic fracturing fluids or other subsurface fluids to move to drinking water resources. There have been cases in which hydraulic fracturing at one well has affected a nearby oil and gas well or its fracture network, resulting in unexpected pressure increases at the nearby well, damage to the nearby well, or spills at the surface of the nearby well. These well communication events, or “frac hits,”

[1] Section 6.3.2 in Chapter 6.

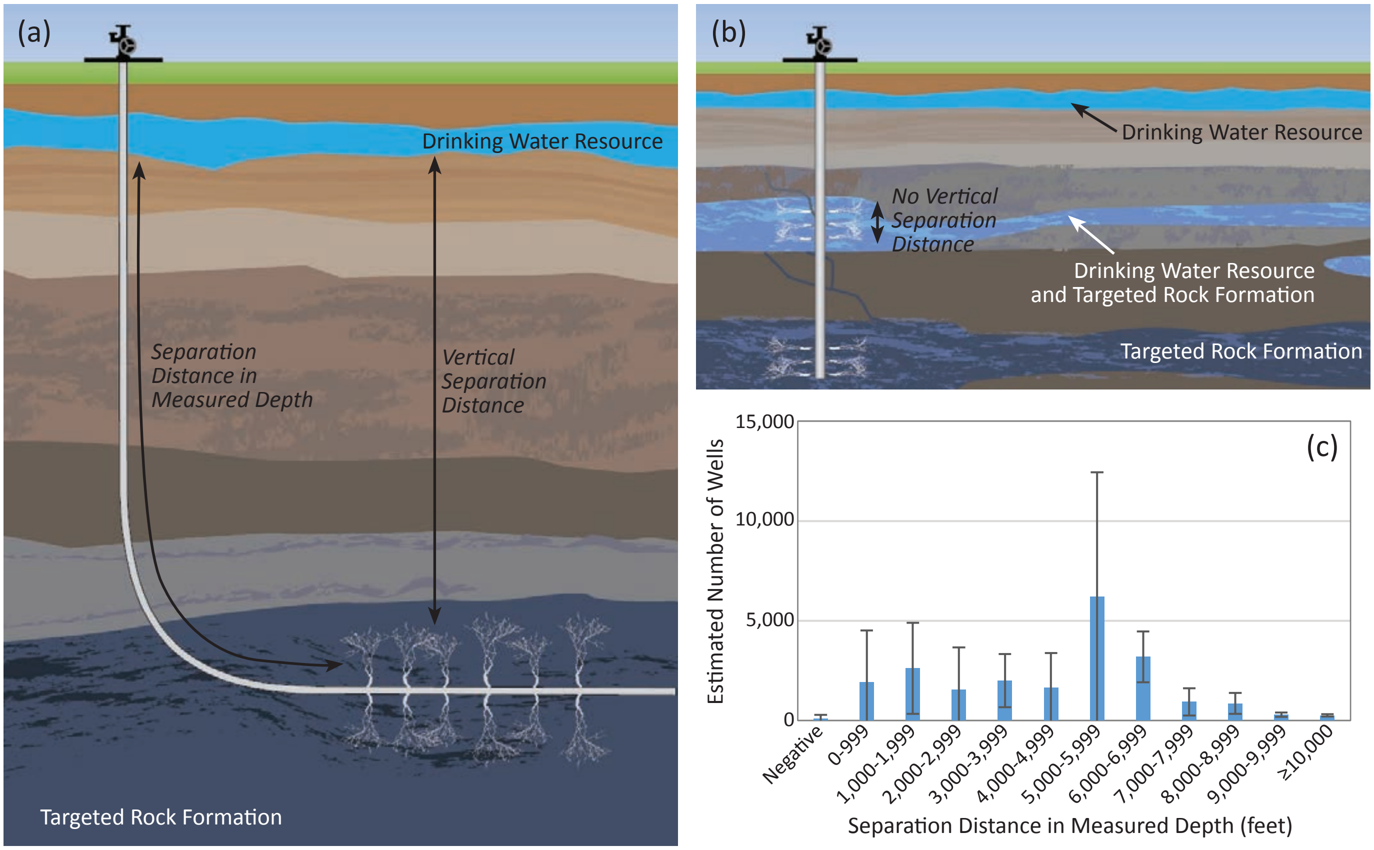


Figure ES-7. Examples of different subsurface environments in which hydraulic fracturing takes place. In panel a, there are thousands of feet between the base of the underground drinking water resource and the part of the well that is hydraulically fractured. Panel b illustrates the co-location of ground water and oil and gas resources. In these types of situations, there is no separation between the shallowest point of hydraulic fracturing within the well and the bottom of the underground drinking water resource. Panel c shows the estimated distribution of separation distances for approximately 23,000 oil and gas production wells hydraulically fractured by nine service companies between 2009 and 2019 (U.S. EPA, 2015d). The separation distance is the distance along the well between the point of shallowest hydraulic fracturing in the well and the base of the protected groundwater resource (illustrated in panel a). The error bars in panel c display 95% confidence intervals.

have been reported in New Mexico, Oklahoma, and other locations. Based on the available information, frac hits most commonly occur when multiple wells are drilled from the same surface location and when wells are spaced less than 1,100 feet (335 meters) apart. Frac hits have also been observed at wells up to 8,422 feet (2,567 meters) away from a well undergoing hydraulic fracturing.

Abandoned wells near a well undergoing hydraulic fracturing can provide a pathway for vertical fluid movement to drinking water resources if those wells were not properly plugged or if the plugs and cement have degraded over time. For example, an abandoned well in Pennsylvania produced a 30-foot (9-meter) geyser of brine and gas for more than a week after hydraulic fracturing of a nearby gas well. The potential for fluid movement along abandoned wells may be a significant issue in areas with historic oil and gas exploration and production. Various studies estimate the number of abandoned wells in the United States to be significant. For instance, the Interstate Oil and Gas Compact Commission estimates that over 1 million wells were drilled in the United States prior to the enactment of state oil and gas regulations (IOGCC, 2008). The location and condition of many of these wells are unknown, and some states have programs to find and plug abandoned wells.

Well Injection Conclusions

Impacts on drinking water resources associated with the well injection stage of the hydraulic fracturing water cycle have occurred in some instances. In particular, mechanical integrity failures have allowed gases or liquids to move to underground drinking water resources. Additionally, hydraulic fracturing has occurred within underground drinking water resources in parts of the United States. This practice introduces hydraulic fracturing fluids into underground drinking water resources. Consequently, the mechanical integrity of the well and the vertical separation distance between the targeted rock formation and underground drinking water resources are important factors that affect the frequency and severity of impacts on drinking water resources. The presence of multiple layers of cemented casing and thousands of feet of rock between hydraulically fractured rock formations and underground drinking water resources can reduce the frequency of impacts on drinking water resources during the well injection stage of the hydraulic fracturing water cycle.

Produced Water Handling

The on-site collection and handling of water that returns to the surface after hydraulic fracturing and the transportation of that water for disposal or reuse.

Relationship to Drinking Water Resources

Spills of produced water can reach groundwater and surface water resources.

After hydraulic fracturing, the injection pressure applied to the oil or gas production well is released, and the direction of fluid flow reverses, causing fluid to flow out of the well. The fluid that initially returns to the surface after hydraulic fracturing is mostly hydraulic fracturing fluid and is sometimes called "flowback" (Text Box ES-9). As time goes on, the fluid that returns to the surface contains water and economic quantities of oil and/or gas that are separated and collected. Water that returns to the surface during oil and gas production is similar in composition to the fluid naturally found in the targeted rock formation and is typically called "produced water." The term "produced water" is also used to refer to any water, including flowback, that returns to the surface through the production well as a by-product of oil and gas production. This latter definition of "produced water" is used in this report.

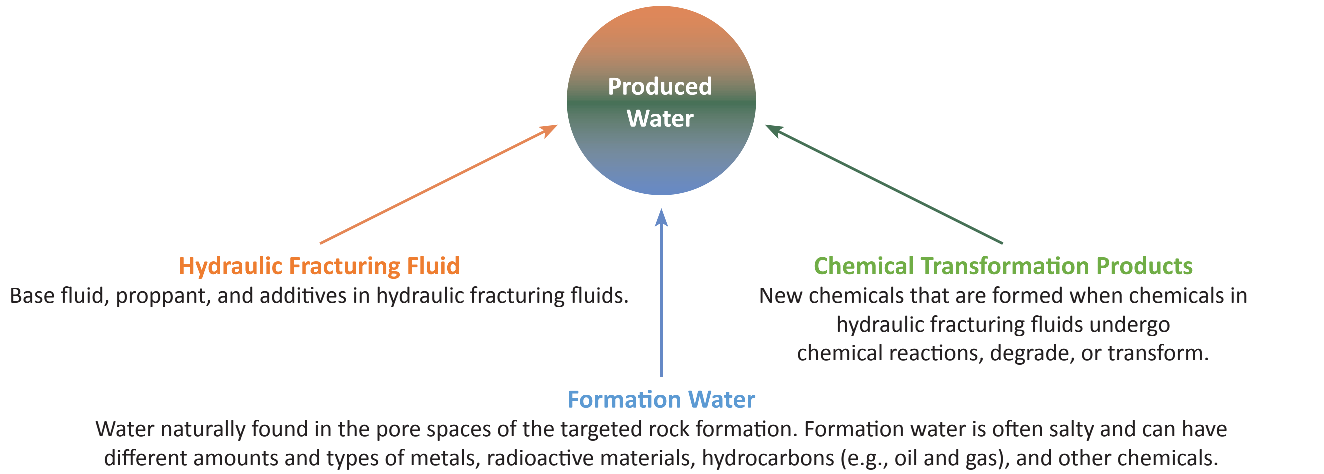
Produced water can contain many constituents, depending on the composition of the injected hydraulic fracturing fluid and the type of rock hydraulically fractured. Knowledge of the chemical composition of produced water comes from the collection and analysis of produced water samples, which often requires advanced laboratory equipment and techniques that can detect and quantify chemicals in produced water. In general, produced water has been found to contain:

- Salts, including those composed from chloride, bromide, sulfate, sodium, magnesium, and calcium;
- Metals, including barium, manganese, iron, and strontium;
- Naturally-occurring organic compounds, including benzene, toluene, ethylbenzene, xylenes (BTEX), and oil and grease;
- Radioactive materials, including radium; and
- Hydraulic fracturing chemicals and their chemical transformation products.

The amount of these constituents in produced water varies across the United States, both within

Text Box ES-9: Produced Water from Hydraulically Fractured Oil and Gas Production Wells

Water of varying quality is a byproduct of oil and gas production. The composition and volume of produced water varies by well, rock formation, and time after hydraulic fracturing. Produced water can contain hydraulic fracturing fluid, formation water, and chemical transformation products.

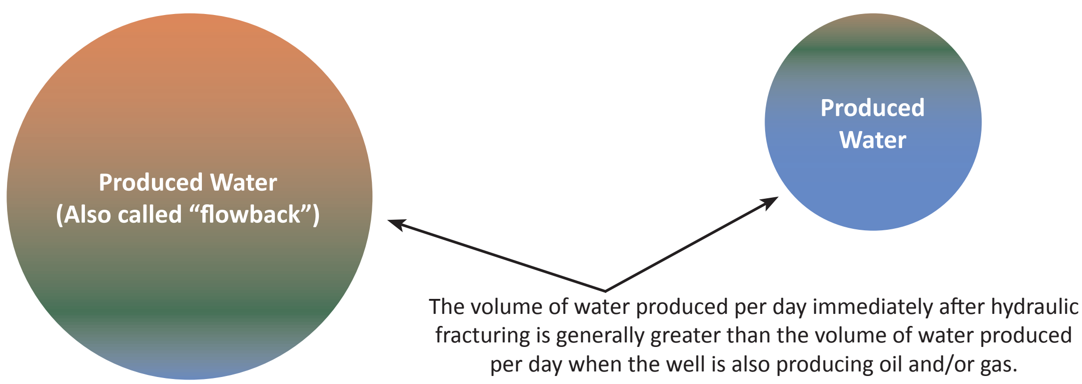


Water Produced Immediately After Hydraulic Fracturing

Generally, the fluid that initially returns to the surface is mostly a mixture of the injected hydraulic fracturing fluid and its reaction and degradation products.

Water Produced During Oil or Gas Production

The fluid that returns to the surface when oil and/or gas is produced generally resembles the formation water.



and among different rock formations. Produced water from shale and tight gas formations is typically very salty compared to produced water from coalbed methane formations. For example, the salinity of produced water from the Marcellus Shale has been reported to range from less than 1,500 milligrams per liter (mg/L) of total dissolved solids to over 300,000 mg/L, while produced water from coalbed methane formations has been reported to range from 170 mg/L of total dissolved solids to nearly 43,000 mg/L.[1] Shale and sandstone formations also commonly contain radioactive materials, including uranium, thorium, and radium. As a result, radioactive materials have been detected in produced water from these formations.

Produced water volumes can vary by well, rock formation, and time after hydraulic fracturing. Vol-

[1] For comparison, the average salinity of seawater is approximately 35,000 mg/L of total dissolved solids.

umes are often described in terms of the volume of hydraulic fracturing fluid used to fracture the well. For example, Figure ES-4 shows that wells in the Marcellus Shale typically produce 10-30% of the volume injected in the first 10 years after hydraulic fracturing. In comparison, some wells in the Barnett Shale have produced 100% of the volume injected in the first three years.

Because of the large volumes used for hydraulic fracturing [about 4 million gallons (15 million liters) per well in the Marcellus Shale and the Barnett Shale], hundreds of thousands to millions of gallons of produced water need to be collected and handled at the well site. The volume of water produced per day generally decreases with time, so the volumes handled on site immediately after hydraulic fracturing can be much larger than the volumes handled when the well is producing oil and/or gas (Text Box ES-9).

Produced water flows from the well to on-site tanks or pits through a series of pipes or flowlines (Text Box ES-10) before being transported offsite via trucks or pipelines for disposal or reuse. While produced water collection, storage, and transportation systems are designed to contain produced water, spills can occur. Changes in drinking water quality can occur if produced water spills reach groundwater or surface water resources.

Produced water spills have been reported across the United States. Median spill volumes among the datasets reviewed for this report ranged from approximately 340 gallons (1,300 liters) to 1,000 gallons (3,800 liters) per spill.[1] There were, however, a small number of large volume spills. In North Dakota, for example, there were 12 spills greater than 21,000 gallons (79,500 liters), five spills greater than 42,000 gallons (160,000 liters), and one spill of 2.9 million gallons (11 million liters) in 2015. Common causes of produced water spills included human error and equipment leaks or failures. Common sources of produced water spills included hoses or lines and storage equipment.

Spills of produced water have reached groundwater and surface water resources. In U.S. EPA (2015c), 30 of the 225 (13%) produced water spills characterized were reported to have reached surface water (e.g., creeks, ponds, or wetlands), and one was reported to have reached groundwater. Of the spills that were reported to have reached surface water, reported spill volumes ranged from less than 170 gallons (640 liters) to almost 74,000 gallons (280,000 liters). A separate assessment of produced water spills reported to the California Office of Emergency Services between January 2009 and December 2014 reported that 18% of the spills impacted waterways (CCST, 2015).

Documented cases of water resource impacts from produced water spills provide insights into the types of impacts that can occur. In most of the cases reviewed for this report, documented impacts included elevated levels of salinity in groundwater and/or surface water resources.[2] For example, the largest produced water spill reported in this report occurred in North Dakota in 2015, when approximately 2.9 million gallons (11 million liters) of produced water spilled from a broken pipeline. The spilled fluid flowed into Blacktail Creek and increased the concentration of chloride and the electrical conductivity of the creek; these observations are consistent with an increase in water salinity. Elevated levels of electrical conductivity and chloride were also found downstream in the Little Muddy River and the Missouri River. In another example, pits holding flowback fluids overflowed in Kentucky in 2007. The spilled fluid reached the Acorn Fork Creek, decreasing the pH of the creek and increasing the electrical conductivity.

Site-specific studies of historical produced water releases highlight the role of local geology in the movement of produced water through the environ-

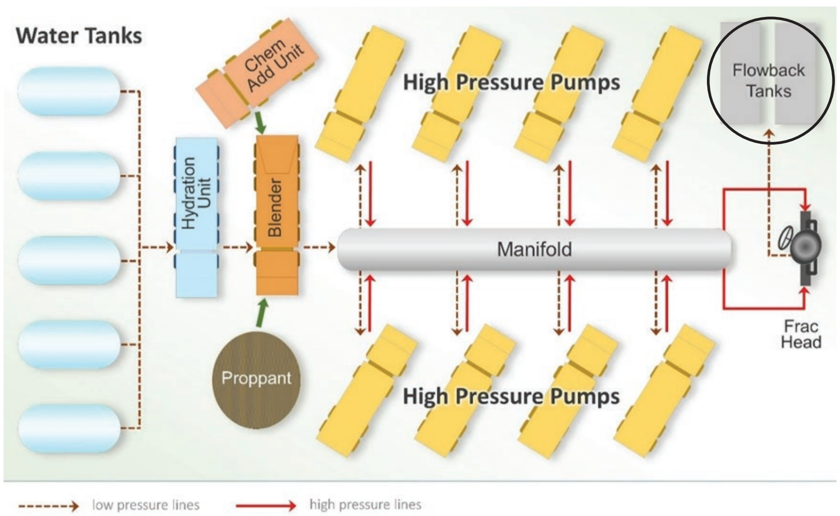
[1] See Section 7.4 in Chapter 7.

[2] Groundwater impacts from produced water management practices are described in Chapter 8 and summarized in the "Wastewater Disposal and Reuse" section below.

Text Box ES-10: On-Site Storage of Produced Water

Water that returns to the surface after hydraulic fracturing is collected and stored on site in pits or tanks.

Above: Flowback pit. (Source: U.S. DOE/NETL)
Right: Flowback tanks. (Source: U.S. EPA)



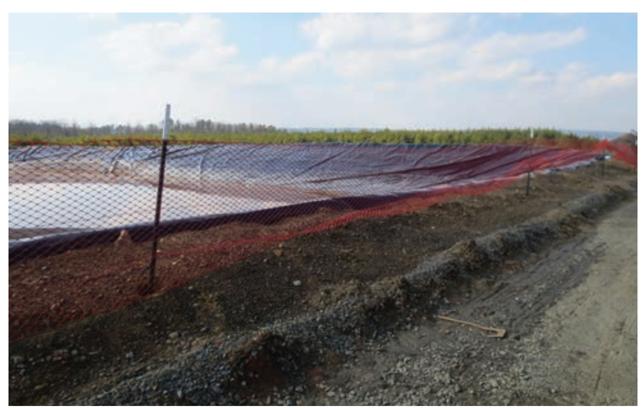
Produced Water Storage Immediately after Hydraulic Fracturing

After hydraulic fracturing, water is returned to the surface. Water initially produced from the well after hydraulic fracturing is sometimes called "flowback." This water can be stored onsite in tanks or pits before being taken offsite for injection in Class II wells, reuse in other hydraulic fracturing operations, or aboveground disposal.

Source: Adapted from Olson (2011) and BJ Services Company (2009)

Produced Water Storage During Oil or Gas Production

Water is generally produced throughout the life of an oil and gas production well. During oil and gas production, the equipment on the well pad often includes the wellhead and storage tanks or pits for gas, oil, and produced water.



Above: Produced water storage pit. (Source: U.S. EPA)
Left: Produced water storage tanks. (Source: U.S. EPA)

ment. Whittemore (2007) described a site in Kansas where low permeability soils and rock caused produced water to primarily flow over the land surface to nearby surface water resources, reducing the amount of produced water that infiltrated soil. In contrast, Otton et al. (2007) explored the release of produced water and oil from two pits in Oklahoma. In this case, produced water from the pits flowed through thin soil and into the underlying, permeable rock. Produced water was also identified in deeper, less permeable rock. The authors suggest that produced water moved into the deeper, less permeable rock through natural fractures. Together, these studies highlight the role of preferential flow paths (i.e., paths of least resistance) in the movement of produced water through the environment.

Spill response activities likely reduce the severity of impacts on groundwater and surface water resources from produced water spills. For example, in the North Dakota example noted above, absorbent booms were placed in the affected creek and contaminated soil and oil-coated ice were removed from the site. In another example, a pipeline leak in Pennsylvania spilled approximately 11,000 gallons (42,000 liters) of produced water, which flowed into a nearby stream. In response, the pipeline was shut off, a dam was constructed to contain the spilled produced water, water was removed from the stream, and the stream was flushed with fresh water. In both examples, it was not possible to quantify how spill response activities reduced the severity of impacts on groundwater or surface water resources. However, actions taken after the spills were designed to stop produced water from entering the environment (e.g., shutting off a pipeline), remove produced water from the environment (e.g., using absorbent booms), and reduce the concentration of produced water constituents introduced into water resources (e.g., flushing a stream with fresh water).

The severity of impacts on water quality from spills of produced water depends on the identity and amount of produced water constituents that reach groundwater or surface water resources, the toxicity of those constituents, and the characteristics of the receiving water resource.[1] In particular, spills of produced water can have high levels of total dissolved solids, which affects how the spilled fluid moves through the environment. When a spilled fluid has greater levels of total dissolved solids than groundwater, the higher-density fluid can move downward through groundwater resources. Depending on the flow rate and other properties of the groundwater resource, impacts from produced water spills can last for years.

Produced Water Handling Conclusions

Spills of produced water during the produced water handling stage of the hydraulic fracturing water cycle have reached groundwater and surface water resources in some cases. Several cases of water resource impacts from produced water spills suggest that impacts are characterized by increases in the salinity of the affected groundwater or surface water resource. In the absence of direct pathways to groundwater resources (e.g., fractured rock), large volume spills are more likely to travel further from the site of the spill, potentially to groundwater or surface water resources. Additionally, saline produced water can migrate downward through soil and into groundwater resources, leading to longer-term groundwater contamination. Spill prevention and response activities can prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids.

[1] Human health hazards associated with chemicals detected in produced water are discussed in Chapter 9 and summarized in the “Chemicals in the Hydraulic Fracturing Water Cycle” section below.

Wastewater Disposal and Reuse

The disposal and reuse of hydraulic fracturing wastewater.

Relationship to Drinking Water Resources

Disposal practices can release inadequately treated or untreated hydraulic fracturing wastewater to groundwater and surface water resources.

In general, produced water from hydraulically fractured oil and gas production wells is managed through injection in Class II wells, reuse in other hydraulic fracturing operations, or various aboveground disposal practices (Text Box ES-11). In this report, produced water from hydraulically fractured oil and gas wells that is being managed through one of the above management strategies is referred to as "hydraulic fracturing wastewater." Wastewater management choices are affected by cost and other factors, including: the local availability of disposal methods; the quality of produced water; the volume, duration, and flow rate of produced water; federal, state, and local regulations; and well operator preferences.

Available information suggests that hydraulic fracturing wastewater is mostly managed through injection in Class II wells. Veil (2015) estimated that 93% of produced water from the oil and gas industry was injected in Class II wells in 2012. Although this estimate included produced water from oil and gas wells in general, it is likely indicative of nationwide management practices for hydraulic fracturing wastewater. Disposal of hydraulic fracturing wastewater in Class II wells is often cost-effective, especially when a Class II disposal well is located within a reasonable distance from a hydraulically fractured oil or gas production well. In particular, large numbers of active Class II disposal wells are found in Texas (7,876), Kansas (5,516), Oklahoma (3,837), Louisiana (2,448), and Illinois (1,054) (U.S. EPA, 2016). Disposal of hydraulic fracturing wastewater in Class II wells has been associated with earthquakes in several states, which may reduce the availability of injection in Class II wells as a wastewater disposal option in these states.

Nationwide, aboveground disposal and reuse of hydraulic fracturing wastewater are currently practiced to a much lesser extent compared to injection in Class II wells, and these management strategies appear to be concentrated in certain parts of the United States. For example, approximately 90% of hydraulic fracturing wastewater from Marcellus Shale gas wells in Pennsylvania was reused in other hydraulic fracturing operations in 2013 (Figure ES-4a). Reuse in hydraulic fracturing operations is practiced in some other areas of the United States as well, but at lower rates (approximately 5-20%). Evaporation ponds and percolation pits have historically been used in the western United States to manage produced water from the oil and gas industry and have likely been used to manage hydraulic fracturing wastewater. Percolation pits, in particular, were commonly reported to have been used to manage produced water from stimulated wells in Kern County, California, between 2011 and 2014.[1] Beneficial uses (e.g., livestock watering and irrigation) are also practiced in the western United States if the water quality is considered acceptable, although available data on the use of these practices are incomplete.

Aboveground disposal practices generally release treated or, under certain conditions, untreated wastewater directly to surface water or the land surface (e.g., wastewater treatment facilities, evaporation pits, or irrigation). If released to the land surface,

[1] Hydraulic fracturing was the predominant stimulation practice. Other stimulation practices included acid fracturing and matrix acidizing. California updated its regulations in 2015 to prohibit the use of percolation pits for the disposal of fluids produced from stimulated wells.

Text Box ES-11: Hydraulic Fracturing Wastewater Management

Produced water from hydraulically fractured oil and gas production wells is often, but not always, considered a waste product to be managed. Hydraulic fracturing wastewater (i.e., produced water from hydraulically fractured wells) is generally managed through injection in Class II wells, reuse in other hydraulic fracturing operations, and various aboveground disposal practices.

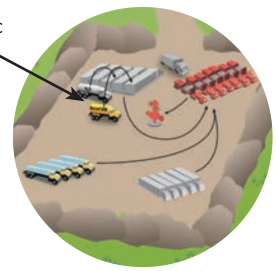
Injection in Class II Wells

Most oil and gas wastewater—including hydraulic fracturing wastewater—is injected in Class II wells, which are regulated under the Underground Injection Control Program of the Safe Drinking Water Act.

Class II wells are used to inject wastewater associated with oil and gas production underground. Fluids can be injected for disposal or to enhance oil or gas production from nearby oil and gas production wells.

Reuse in Other Hydraulic Fracturing Operations

Hydraulic fracturing wastewater can be used, in combination with fresh water, to make up hydraulic fracturing fluids at nearby hydraulic fracturing operations.

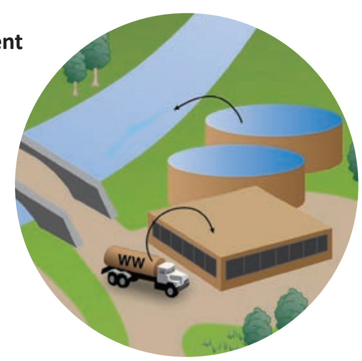


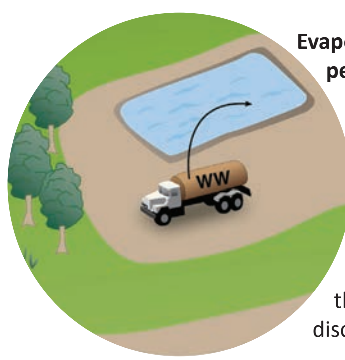
Reuse in other hydraulic fracturing operations depends on the quality and quantity of the available wastewater, the cost associated with treatment and transportation of the wastewater, and local water demand for hydraulic fracturing.

Aboveground Disposal Practices

Aboveground disposal of treated and untreated hydraulic fracturing wastewater can take many forms, including release to surface water resources and land application.

Some **wastewater treatment facilities** treat hydraulic fracturing wastewater and release the treated wastewater to surface water. Solid or liquid by-products of the treatment process can be sent to landfills or injected underground.





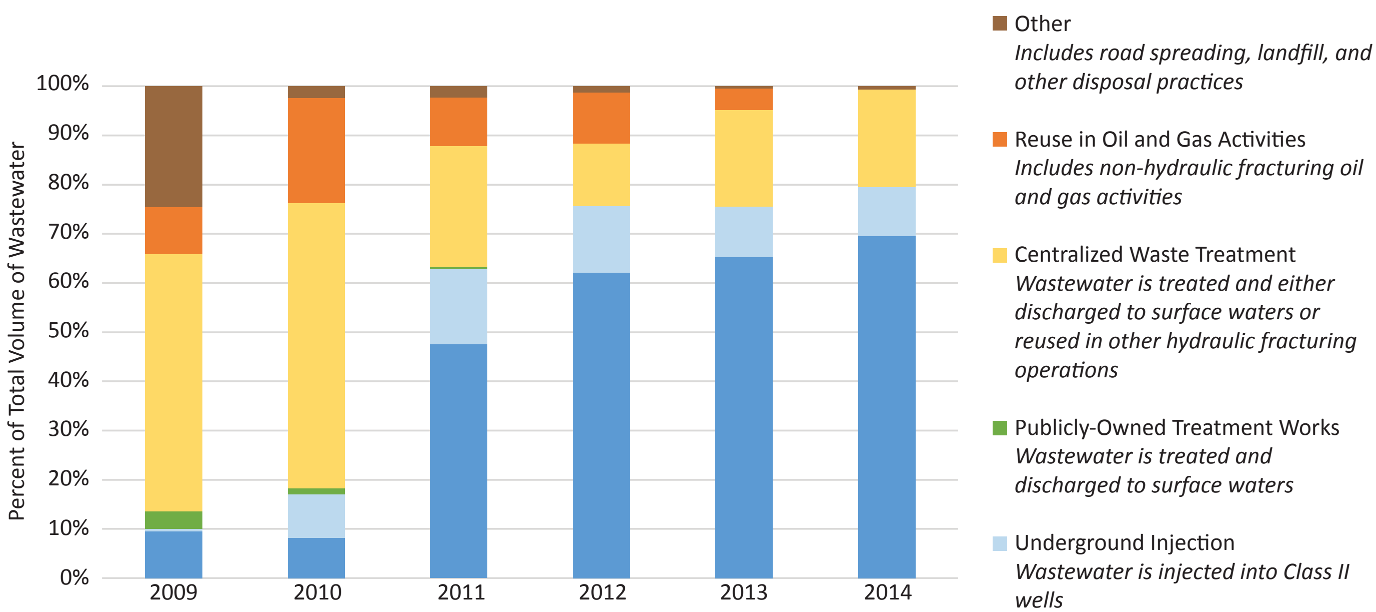
Evaporation ponds and **percolation pits** can be used for hydraulic fracturing wastewater disposal. Evaporation ponds allow liquid waste to naturally evaporate. Percolation pits allow wastewater to move into the ground, although this practice has been discontinued in most states.

Federal and state regulations affect aboveground disposal management options. For example, existing federal regulations generally prevent the direct release of wastewater pollutants to waters of the United States from onshore oil and gas extraction facilities east of the 98th meridian. However, in the arid western portion of the continental United States (west of the 98th meridian), direct discharges of wastewater from onshore oil and gas extraction facilities to waters of the United States may be permitted if the produced water has a use in agriculture or wildlife propagation and meets established water quality criteria when discharged.

treated or untreated wastewater can move through soil to groundwater resources. Because the ultimate fate of the wastewater can be groundwater or surface water resources, the aboveground disposal of hydraulic fracturing wastewater, in particular, can impact drinking water resources.

Impacts on drinking water resources from the aboveground disposal of hydraulic fracturing wastewater have been documented. For example, early wastewater management practices in the Marcellus Shale region in Pennsylvania included the use of wastewater treatment facilities that released (i.e., discharged) treated wastewater to surface waters (Figure ES-8). The wastewater treatment facilities were unable to adequately remove the high levels of total dissolved solids found in produced water from Marcellus Shale gas wells, and the discharges contributed to elevated levels of total dissolved solids (particularly bromide) in the Monongahela River Basin. In the Allegheny River Basin, elevated bromide levels were linked to increases in the concentration of hazardous disinfection byproducts in at least one downstream drinking water facility and a shift to more toxic brominated disinfection byproducts.[1] In response, the Pennsylvania Department of Environmental Protection revised existing regulations to prevent these discharges and also requested that oil and gas operators voluntarily stop bringing certain kinds of hydraulic fracturing wastewater to facilities that discharge inadequately treated wastewater to surface waters.[2]

The scientific literature and recent data from the Pennsylvania Department of Environmental Protection suggest that other produced water constituents



Data from the Pennsylvania Department of Environmental Protection (2015).

Figure ES-8. Changes in wastewater management practices over time in the Marcellus Shale area of Pennsylvania.

[1] Disinfection byproducts form through chemical reactions between organic material and disinfectants, which are used in drinking water treatment. Human health hazards associated with disinfection byproducts are described in Section 9.5.6 in Chapter 9.

[2] See Text Box 8-1 in Chapter 8.

(e.g., barium, strontium, and radium) may have been introduced to surface waters through the release of inadequately treated hydraulic fracturing wastewater. In particular, radium has been detected in stream sediments at or near wastewater treatment facilities that discharged inadequately treated hydraulic fracturing wastewater. Such sediments can migrate if they are disturbed during dredging or flood events. Additionally, residuals from the treatment of hydraulic fracturing wastewater (i.e., the solids or liquids that remain after treatment) are concentrated in the constituents removed during treatment, and these residuals can impact groundwater or surface water resources if they are not managed properly.

Impacts on groundwater and surface water resources from current and historic uses of lined and unlined pits, including percolation pits, in the oil and gas industry have been documented. For example, Kell (2011) reported 63 incidents of non-public water supply contamination from unlined or inadequately constructed pits in Ohio between 1983 and 2007, and 57 incidents of groundwater contamination from unlined produced water disposal pits in Texas prior to 1984. Other cases of impacts have been identified in several states, including New Mexico, Oklahoma, Pennsylvania, and Wyoming.[1] Impacts among these cases included the detection of volatile organic compounds in groundwater resources, wastewater reaching surface water resources from pit overflows, and wastewater reaching groundwater resources through liner failures. Based on documented impacts on groundwater resources from unlined pits, many states have implemented regulations that prohibit percolation pits or unlined storage pits for either hydraulic fracturing wastewater or oil and gas wastewater in general.

The severity of impacts on drinking water resources from the aboveground disposal of hydraulic fracturing wastewater depends on the volume and quality of the discharged wastewater and the characteristics of the receiving water resource. In general, large surface water resources with high flow rates can reduce the severity of impacts through dilution, although impacts may not be eliminated. In contrast, groundwater is generally slow moving, which can lead to an accumulation of hydraulic fracturing wastewater contaminants in groundwater from continuous or repeated discharges to the land surface; the resulting contamination can be long-lasting. The severity of impacts on groundwater resources will also be influenced by soil and sediment properties and other factors that control the movement or degradation of wastewater constituents.

Wastewater Disposal and Reuse Conclusions

The aboveground disposal of hydraulic fracturing wastewater has impacted the quality of groundwater and surface water resources in some instances. In particular, discharges of inadequately treated hydraulic fracturing wastewater to surface water resources have contributed to elevated levels of hazardous disinfection byproducts in at least one downstream drinking water system. Additionally, the use of lined and unlined pits for the storage or disposal of oil and gas wastewater has impacted surface and groundwater resources. Unlined pits, in particular, provide a direct pathway for contaminants to reach groundwater. Wastewater management is dynamic, and recent changes in state regulations and practices have been made to limit impacts on groundwater and surface water resources from the aboveground disposal of hydraulic fracturing wastewater.

[1] See Section 8.4.5 in Chapter 8.

Chemicals in the Hydraulic Fracturing Water Cycle

Chemicals are present in the hydraulic fracturing water cycle. During the chemical mixing stage of the hydraulic fracturing water cycle, chemicals are intentionally added to water to alter its properties for hydraulic fracturing (Text Box ES-6). Produced water, which is collected, handled, and managed in the last two stages of the hydraulic fracturing water cycle, contains chemicals added to hydraulic fracturing fluids, naturally occurring chemicals found in hydraulically fractured rock formations, and any chemical transformation products (Text Box ES-9). By evaluating available data sources, we compiled a list of 1,606 chemicals that are associated with the hydraulic fracturing water cycle, including 1,084 chemicals reported to have been used in hydraulic fracturing fluids and 599 chemicals detected in produced water. This list represents a national analysis; an individual well would likely have a fraction of the chemicals on this list and may have other chemicals that were not included on this list.

In many stages of the hydraulic fracturing water cycle, the severity of impacts on drinking water resources depends, in part, on the identity and amount of chemicals that enter the environment. The properties of a chemical influence how it moves and transforms in the environment and how it interacts with the human body. Therefore, some chemicals in the hydraulic fracturing water cycle are of more concern than others because they are more likely to move with water (e.g., spilled hydraulic fracturing fluid) to drinking water resources, persist in the environment (e.g., chemicals that do not degrade), and/or affect human health.

Evaluating potential hazards from chemicals in the hydraulic fracturing water cycle is most useful at local and/or regional scales because chemical use for hydraulic fracturing can vary from well to well and because the characteristics of produced water are influenced by the geochemistry of hydraulically fractured rock formations. Additionally, site-specific characteristics (e.g., the local landscape, and soil and subsurface permeability) can affect whether and how chemicals enter drinking water resources, which influences how long people may be exposed to specific chemicals and at what concentrations. As a first step for informing site-specific risk assessments, the EPA compiled toxicity values for chemicals in the hydraulic fracturing water cycle from federal, state, and international sources that met the EPA's criteria for inclusion in this report.[1,2]

The EPA was able to identify chronic oral toxicity values from the selected data sources for 98 of the 1,084 chemicals that were reported to have been used in hydraulic fracturing fluids between 2005 and 2013. Potential human health hazards associated with chronic oral exposure to these chemicals include cancer, immune system effects, changes in body weight, changes in blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity. Of the chemicals most frequently reported to FracFocus 1.0, nine had toxicity values from the selected data sources (Table ES-3). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, developmental toxicity (extra cervical ribs), reproductive toxicity, and decreased terminal body weight.

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[1] Specifically, the EPA compiled noncancer oral reference values and cancer oral slope factors (Chapter 9). A reference value describes the dose of a chemical that is likely to be without an appreciable risk of adverse health effects. In the context of this report, the term "reference value" generally refers to reference values for noncancer effects occurring via the oral route of exposure and for chronic durations. An oral slope factor is an upper-bound estimate on the increased cancer risk from a lifetime oral exposure to an agent.

[2] The EPA's criteria for inclusion in this report are described in Section 9.4.1 in Chapter 9. Sources of information that met these criteria are listed in Table 9-1 of Chapter 9.

Table ES-3. Available chronic oral reference values for hydraulic fracturing chemicals reported in 10% or more of disclosures in FracFocus 1.0.

Chemical Name (CASRN)[a]	Chronic Oral Reference Value (milligrams per kilogram per day)	Critical Effect	Percent of FracFocus 1.0 Disclosures[b]
Propargyl alcohol (107-19-7)	0.002[c]	Renal and hepatotoxicity	33
1,2,4-Trimethylbenzene (95-63-6)	0.01[c]	Decreased pain sensitivity	13
Naphthalene (91-20-3)	0.02[c]	Decreased terminal body weight	19
Sodium chlorite (7758-19-2)	0.03[c]	Neuro-developmental effects	11
2-Butoxyethanol (111-76-2)	0.1[c]	Hemosiderin deposition in the liver	23
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (68424-85-1)	0.44[d]	Decreased body weight and weight gain	12
Formic acid (64-18-6)	0.9[e]	Reproductive toxicity	11
Ethylene glycol (107-21-1)	2[c]	Kidney toxicity	47
Methanol (67-56-1)	2[c]	Extra cervical ribs	73

[a]"Chemical" refers to chemical substances with a single CASRN; these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates).
[b]Analysis considered 35,957 disclosures that met selected quality assurance criteria. See Table 9-2 in Chapter 9.
[c]From the EPA Integrated Risk Information System database.
[d]From the EPA Human Health Benchmarks for Pesticides database.
[e]From the EPA Provisional Peer-Reviewed Toxicity Value database.

Chronic oral toxicity values from the selected data sources were identified for 120 of the 599 chemicals detected in produced water. Potential human health hazards associated with chronic oral exposure to these chemicals include liver toxicity, kidney toxicity, neurotoxicity, reproductive and developmental toxicity, and carcinogenesis. Chemical-specific toxicity values are included in Chapter 9.

Chemicals in the Hydraulic Fracturing Water Cycle Conclusions

Some of the chemicals in the hydraulic fracturing water cycle are known to be hazardous to human health. Of the 1,606 chemicals identified by the EPA, 173 had chronic oral toxicity values from federal, state, and international sources that met the EPA's criteria for inclusion in this report. These data alone, however, are insufficient to determine which chemicals have the greatest potential to impact drinking water resources and human health. To understand whether specific chemicals can affect human health through their presence in drinking water, data on chemical concentrations in drinking water would be needed. In the absence of these data, relative hazard potential assessments could be conducted at local and/or regional scales using the multi-criteria decision analysis approach outlined in Chapter 9. This approach combines available chemical occurrence data with selected chemical, physical, and toxicological properties to place the severity of potential impacts (i.e., the toxicity of specific chemicals) into the context of factors that affect the likelihood of impacts (i.e., frequency of use, and chemical and physical properties relevant to environmental fate and transport).

Data Gaps and Uncertainties

The information reviewed for this report included cases of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. Using these cases and other data, information, and analyses, we were able to identify factors that likely result in more frequent or more severe impacts on drinking water resources. However, there were instances in which we were unable to form conclusions about the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources and/or the factors that influence the frequency or severity of impacts. Below, we provide perspective on the data gaps and uncertainties that prevented us from drawing additional conclusions about the potential for impacts on drinking water resources and/or the factors that affect the frequency and severity of impacts.

In general, comprehensive information on the location of activities in the hydraulic fracturing water cycle is lacking, either because it is not collected, not publicly available, or prohibitively difficult to aggregate. This includes information on the:

- Above- and belowground locations of water withdrawals for hydraulic fracturing;
- Surface locations of hydraulically fractured oil and gas production wells, where the chemical mixing, well injection, and produced water handling stages of the hydraulic fracturing water cycle take place;
- Belowground locations of hydraulic fracturing, including data on fracture growth; and
- Locations of hydraulic fracturing wastewater management practices, including the disposal of treatment residuals.

There can also be uncertainty in the location of drinking water resources. In particular, depths of groundwater resources that are, or in the future could be, used for drinking water are not always known. If comprehensive data about the locations of both drinking water resources and activities in the hydraulic fracturing water cycle were available, it would have been possible to more completely identify areas in the United States in which hydraulic fracturing-related activities either directly interact with drinking water resources or have the potential to interact with drinking water resources.

In places where we know activities in the hydraulic fracturing water cycle have occurred or are occurring, data that could be used to characterize the presence, migration, or transformation of hydraulic fracturing-related chemicals in the environment before, during, and after hydraulic fracturing were scarce. Specifically, local water quality data needed to compare pre- and post-hydraulic fracturing conditions are not usually collected or readily available. The limited amount of data collected before, during, and after activities in the hydraulic fracturing water cycle reduces the ability to determine whether these activities affected drinking water resources.

Site-specific cases of alleged impacts on underground drinking water resources during the well injection stage of the hydraulic fracturing water cycle are particularly challenging to understand (e.g., methane migration in Dimock, Pennsylvania; the Raton Basin of Colorado; and Parker County, Texas[1]). This is because the subsurface environment is complex and belowground fluid movement is not directly observable. In cases of alleged impacts, activities in the hydraulic fracturing water cycle may be one of several causes of impacts, including other oil and gas activities, other industries, and natural processes. Thorough scientific investigations are often necessary to narrow down the list of potential causes to a single source at site-specific cases of alleged impacts.

Additionally, information on chemicals in the hydraulic fracturing water cycle (e.g., chemical iden-

[1] See Text Boxes 6-2 (Dimock, Pennsylvania), 6-3 (Raton Basin), and 6-4 (Parker County, Texas) in Chapter 6.

tity; frequency of use or occurrence; and physical, chemical, and toxicological properties) is not complete. Well operators claimed at least one chemical as confidential at more than 70% of wells reported to FracFocus 1.0 (U.S. EPA, 2015a).[1] The identity and concentration of these chemicals, their transformation products, and chemicals in produced water would be needed to characterize how chemicals associated with hydraulic fracturing activities move through the environment and interact with the human body. Identifying chemicals in the hydraulic fracturing water cycle also informs decisions about which chemicals would be appropriate to test for when establishing pre-hydraulic fracturing baseline conditions and in the event of a suspected drinking water impact.

Of the 1,606 chemicals identified by the EPA in hydraulic fracturing fluid and/or produced water, 173 had toxicity values from sources that met the EPA's criteria for inclusion in this report. Toxicity values from these selected data sources were not available for 1,433 (89%) of the chemicals, although many of these chemicals have toxicity data available from other data sources.[2] Given the large number of chemicals identified in the hydraulic fracturing water cycle, this missing information represents a significant data gap that makes it difficult to fully understand the severity of potential impacts on drinking water resources.

Because of the significant data gaps and uncertainties in the available data, it was not possible to fully characterize the severity of impacts, nor was it possible to calculate or estimate the national frequency of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. We were, however, able to estimate impact frequencies in some, limited cases (i.e., spills of hydraulic fracturing fluids or produced water and mechanical integrity failures).[3] The data used to develop these estimates were often limited in geographic scope or otherwise incomplete. Consequently, national estimates of impact frequencies for any stage of the hydraulic fracturing water cycle have a high degree of uncertainty. Our inability to quantitatively determine a national impact frequency or to characterize the severity of impacts, however, did not prevent us from qualitatively describing factors that affect the frequency or severity of impacts at the local level.

Report Conclusions

This report describes how activities in the hydraulic fracturing water cycle can impact—and have impacted—drinking water resources and the factors that influence the frequency and severity of those impacts. It also describes data gaps and uncertainties that limited our ability to draw additional conclusions about impacts on drinking water resources from activities in the hydraulic fracturing water cycle. Both types of information—what we know and what we do not know—provide stakeholders with scientific information to support future efforts.

The uncertainties and data gaps identified throughout this report can be used to identify future efforts to further our understanding of the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources and the factors that affect the frequency and severity of those impacts. Future efforts could include, for example, groundwater and surface water monitoring in areas with hydraulically fractured oil and gas production wells or tar-

[1] Chemical withholding rates in FracFocus have increased over time. Konschnik and Dayalu (2016) reported that 92% of wells reported in FracFocus 2.0 between approximately March 2011 and April 2015 used at least one chemical that was claimed as confidential.

[2] Chapter 9 describes the availability of data in other data sources. The quality of these data sources was not evaluated as part of this report.

[3] See Chapter 10.

geted research programs to better characterize the environmental fate and transport and human health hazards associated with chemicals in the hydraulic fracturing water cycle. Future efforts could identify additional vulnerabilities or other factors that affect the frequency and/or severity of impacts.

In the near term, decision-makers could focus their attention on the combinations of hydraulic fracturing water cycle activities and local- or regional-scale factors that are more likely than others to result in more frequent or more severe impacts. These include:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;
- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;
- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

The above combinations of activities and factors highlight, in particular, the vulnerability of groundwater resources to activities in the hydraulic fracturing water cycle. By focusing attention on the situations described above, impacts on drinking water resources from activities in the hydraulic fracturing water cycle could be prevented or reduced.

Overall, hydraulic fracturing for oil and gas is a practice that continues to evolve. Evaluating the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources will need to keep pace with emerging technologies and new scientific studies. This report provides a foundation for these efforts, while helping to reduce current vulnerabilities to drinking water resources.

Source: U.S. EPA

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Front cover (top): Illustrations of activities in the hydraulic fracturing water cycle. From left to right: Water Acquisition, Chemical Mixing, Well Injection, Produced Water Handling, and Wastewater Disposal and Reuse.

Front cover (bottom): Aerial photographs of hydraulic fracturing activities. Left: Near Williston, North Dakota. Image ©J Henry Fair / Flights provided by LightHawk. Right: Springville Township, Pennsylvania. Image ©J Henry Fair / Flights provided by LightHawk.

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