5-2016

Water Use In Shale Energy Extraction: A Watershed-Level Analysis of Water Availability in Marcellus Shale Extraction

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WATER USE IN SHALE ENERGY EXTRACTION: A WATERSHED-LEVEL ANALYSIS OF WATER AVAILABILITY IN MARCELLUS SHALE EXTRACTION

A Thesis in

Geography

by

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Submitted in Partial Fulfillment

of the Requirements

for the Degree of

Master of Science

May 2016

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Abstract

Discourse surrounding hydraulic fracturing is shrill. Proponents tout economic and geopolitical benefits; opponents claim massive negative social and environmental impacts. Scientific evidence indicates that neither narrative is entirely accurate. Water quantity is a key, mostly unanalyzed question that also impacts water quality because a smaller amount of water with a consistent pollutant level will be more polluted. This question has spatial and temporal scale aspects. Studies showing statewide or annual use figures can indicate minimal water usage by hydraulic fracturing relative to the statewide or annual water availability. However, downscaling extraction and availability to the county or watershed level shows dramatically different results. This paper analyzes national data for hydraulic fracturing in the United States from 1947-2010 to determine the watershed-level sustainability of hydraulic fracturing within Pennsylvania. The results of this analysis show that sustainable volumes of water can be allocated to hydraulic fracturing operations in Pennsylvania if authorities at all levels consider ground and surface water regimes and upstream impacts before permitting new well development. The results from this study can help identify watersheds suited (or not) for future development and provide policymakers with a proposed set of seasonal and local criteria to consider when authorizing new development.

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Acknowledgements

This research would not have been possible without the consistent effort of my committee members, Drs. Kirby Calvert, Tess Russo, and Roger Downs. I had only the vaguest idea of the research I wanted to pursue, and the weekly black whiteboard sessions in Dr. Calvert's office helped fill in the gaps until we narrowed down a suitable research question. His patience knew no bounds as he helped me combine two conceptual frameworks into one. Learning hydrology from Dr. Russo provided the physical science background needed to understand the hydrological system far beyond my previous, laughably basic, knowledge. Dr. Downs was and remains a source of inspiration because he taught me what writing is: not simply putting words on paper or the mechanics of correct grammar, but trying to make a particular point or achieve a particular end with a particular audience.

Without the support of the Department of Geography and Environmental Engineering at the United States Military Academy, I would not be at Penn State. I would not have had the chance to write this thesis; rediscover my love of reading, thinking, and writing; or learn many things I didn't know I didn't know. For that, I have COL Andrew Lohman, the head of the Dirt Department, to thank. I hope I am able to repay the trust you showed me when you called to offer me an assistant professor job early one morning in 2013.

Many, many other people influenced this work in ways big and small—Fish with cartography; Nari with political ecology; Fish, Nari, and Elena for babysitting—but by far the most important is my wife Elizabeth. She has been a constant font of calmness, even (especially) when the seemingly never-ending flow of corrections threatened to sweep me away. She did this despite being pregnant with, delivering, and caring for our first child, all while completing her own research. If I can be half the husband and father she is wife and mother…

Chapter 1 Introduction

Since the first oil well was drilled in western Pennsylvania in 1859, oil and gas have become an ever-expanding portion of the energy mix in the United States, combining to provide approximately 28% of the fuel for electricity generation and virtually all transportation fuel (EIA 2016a; EIA 2016b). As demand and production grew in tandem, so did the need to secure new sources of supply. Initial discoveries, like that of Oil Creek in 1859, were relatively easy to extract and required relatively few inputs. These "conventional" oil and gas deposits collected in near-surface "pools," having migrated upward over geologic time from deep organic strata through the intermediate porous geologic media. In order to extract these fuels, the non-permeable "cap rock" atop the pool is punctured with a drill, allowing natural pressure gradients to force the resource to the surface through a well casing. Social values, institutions, and expectations co-evolved with this relatively low energy input and high energy density (Smil 2008, p. 18).

Over time these conventional sources became increasingly difficult to locate. In order to maintain production levels that could meet increasing demand, oil and gas sealed tightly in deep subsurface shale formations became a focus of policy and industry activity. Deposits in these ancient shale formations had been documented as early as 1821 (Rodriguez and Soeder 2015), but their extraction was not economical on a large scale because of the high production costs compared to conventional sources. Steadily-increasing global demand (and therefore price) for oil and gas in the late 1990s and early

2000s changed the economic calculus behind shale oil and gas extraction. The new economics encouraged the large-scale development of hydraulic fracturing,[1] a combination of the long-used process of horizontal drilling with injection of large volumes of water under high pressure. The new[2] technique proved to be a cheap and effective method of extracting the fuels contained in shale, and because of the new, more complicated production techniques required, the resulting shale fuels became known as "unconventional" resources (Scanlon, Reedy, and Nicot 2014; Kaiser 2012).

For a variety of reasons, the United States (US) has become the world leader in the application of horizontal drilling and hydraulic fracturing to produce unconventional fuels.[3] Due to the dramatic increase in the number and productivity of wells drilled using these techniques (Fontenot et al. 2013; Nicot and Scanlon 2012), the US is considered the world's swing producer of oil at higher price points (Krane and Agerton 2015), a position long held by Saudi Arabia and its massive conventional oil fields. In addition to this huge growth in oil production, from 2000-2011 production of natural gas from shale in the United States grew from 0.4 trillion cubic feet (TCF) to 6.8 TCF per year. The latter figure represents approximately 30% of total US natural gas production, a proportion expected to grow to 49% by 2035 (EIA 2012). With this in mind the terms "shale energy revolution" and the "Golden Age of Gas" (IEA 2011) are hardly hyperbolic, though the latter phrase ignores the parallel rise of shale oil production.

[1] Hydraulic fracturing is also sometimes referred to as "high-volume hydraulic fracturing" or in abbreviated form, HVHF.
[2] Though horizontal drilling has existed since 1929 (Altomare et al. 1993) and hydraulic fracturing since 1947 (Gallegos and Varela 2015), they were not combined on a large scale until the late 1990s.
[3] The US is the leader in production of these fuels due to a fortuitous combination of high local demand for fuel, cheap credit, privatization of mineral rights, an experienced domestic drilling and service industry, and an established, liberalized network of pipelines (*The Economist* 2014).

Production of fuels from shale is well-positioned to become a global phenomenon. The US Energy Information Agency (EIA) has identified at least 41 countries with significant shale fuel resources, and has estimated global technically recoverable resources (including the US) to be 345 billion barrels (Bbbl) of shale oil and 7,299 TCF of shale gas (EIA 2013). The International Energy Agency (IEA) places the US a distant second to Russia in shale oil reserves and fourth behind China, Argentina and Algeria in shale gas reserves, indicating the importance and global breadth of the shale energy industry is likely to increase enormously. As the size of the global industry grows, it is critical that the scientific community assess the known costs, benefits, and effects of shale fuel production and synthesize those results to facilitate effective and well-informed resource management decisions.

One of the most important impacts relates to water. Indeed, hydraulic fracturing has forged new connections at the water-energy nexus. The purpose of this thesis is to develop and apply a method to assess the impacts of hydraulic fracturing on surface water availability through a case study of Pennsylvania. A political-industrial ecology lens is applied to help illuminate the social and technical dimensions of the relationship between water supply, water governance, and hydraulic fracturing. By combining the strongest traditions of the component ecologies into a new method of understanding of hydraulic fracturing, the system's place in and impact on the hydrological cycle and the water governance framework of the state will become clear. This thesis proceeds in four sections: (1) a systematic review of scientific literature on the impacts of hydraulic fracturing on the hydrological cycle; (2) a discussion of political-industrial ecology and its advantages for understanding HVHF and the water-energy nexus; (3) data and methods; (4) results and discussion, and will conclude with suggestions for future research.

Chapter 2 Literature Review

Introduction

Conventional fossil fuels are extracted after they have migrated out of their source rock through porous geologic media, and collected in a "pool" which is sealed by a "cap rock." During drilling, the cap rock is punctured and natural pressure gradients force petroleum to the surface. In contrast, shale fuels are fossil fuels extracted directly from their source rock – in this case, shale rock formations buried thousands of feet underground – and are typically lighter oils and natural gas. Horizontal drilling techniques and hydraulic fracturing are required in order to extract these fuels economically. For these reasons, shale fuels are considered to be "unconventional" fossil resources (Scanlon, Reedy, and Nicot 2014; Kaiser 2012), even though fuels have been extracted from shale formations since 1821 (Rodriguez and Soeder 2015); horizontal drilling has existed since 1929 (Altomare et al. 1993); and hydraulic fracturing since 1947 (Gallegos and Varela 2015). However, relatively high costs and low efficiencies compared to conventional fuel extraction prevented use of these techniques at significant scale. High petroleum prices and improvements in the horizontal drilling and hydraulic fracturing processes, especially combining them into a single process around the turn of the 21st century, have dramatically increased the economic viability, and therefore spread, of the process (see Figure 2.1).

Extracting oil and gas using hydraulic fracturing occurs in five steps (see Figures 2.2 and 2.3). First, an aboveground location for the wellheads of multiple horizontal wells, known as a well pad, is established with at least one vertical well. On average, each well pad requires approximately 5-8 acres of cleared land, in addition to the land required for access roads and gathering pipelines. A vertical well similar to those used for conventional oil and gas extraction is drilled and lined with cement. As the drillbit approaches the source formation, it is slowly oriented from vertical to horizontal using

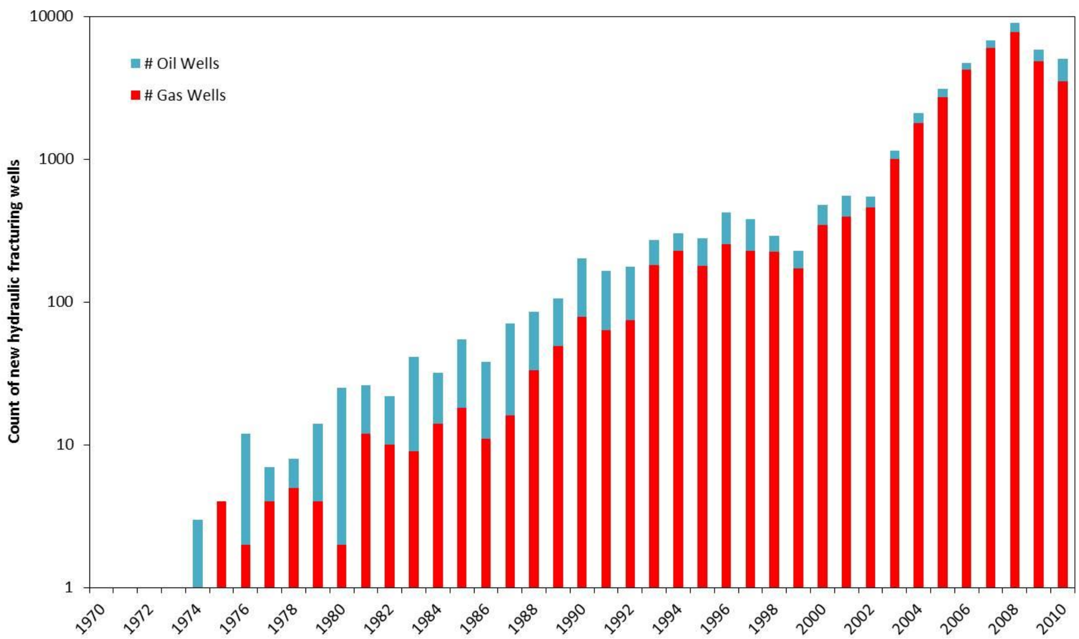

Figure 2.1 The number of hydraulic fracturing wells added in the United States each year from 1970-2010. Red represents the number of wells created for gas and blue represents the number of wells created for oil. Data Source: Gallegos and Varela 2015.

hydraulic pressure in order to create a lateral that cross-sections the shale layer. One or several of these laterals, stretching two or more kilometers into the shale formation, radiate horizontally from each vertical well along different axes, vastly increasing the area of the source formation that can be accessed from one aboveground location (Jackson et al 2014).

After completion of each well, tens of thousands of cubic meters of "fracturing fluid" (a mixture of water, chemicals, and fine silica sand known as "proppant") are injected into the wellbore under high pressure. The wellbore is pressurized by stage, typically a few tens of meters at a time, beginning at the far end of the lateral in order to maintain the required high pressure. This pressure creates a network of fractures in the source rock formation which are held open by the proppants, freeing the resource from within the shale layers. The well is then de-pressurized and the hydrocarbons flow toward the surface along with "flowback" water, which consists mostly of fracturing fluid. Over a period of approximately 10-14 days after depressurization, thousands of cubic meters of flowback returns to the surface (Jiang, Hendrickson, and VanBriesen 2014). Eventually, the ratio of flowback water to fuel is sufficiently low to begin commercial production. At this point, water continues to return to the surface, but is now referred to as "produced water"—i.e., water which has been liberated from the formation by the fracturing process. Produced water flows to the surface throughout the productive life of the well, but at a much slower rate compared to the flowback volumes (a few cubic meters per day rather than a few hundred or a few thousand). The quality and quantity of flowback and produced water are both dependent on the specific basin and the drilling techniques (Benko and Drewes 2008; Nicot et al. 2014). Because produced water has higher salinity and is and more toxic than flowback water (as a result of mixing between deep underground brines and minerals), it must be treated or, as is increasingly the case, recycled back into the process (Mauter et al. 2013; Vidic et al. 2013, SRBC 2014).

In certain cases, a well can be "re-stimulated," or repeatedly fractured, throughout its lifetime to maintain fuel production (Colborn et al. 2011; Jiang, Hendrickson, and VanBriesen 2014). The decision to re-stimulate the well, and the efficacy of this process, is highly dependent on site geology. Although wells have been re-fractured for decades, re-stimulation is exceedingly rare, peaking at 0.35% of wells in 2010 (Gallegos and Varela 2015); however, there is evidence that interest in re-stimulation is growing as a means of increasing yields from existing wells (Bloomberg 2015). For wells that are re-stimulated, the quantities of water required are likely to be similar to the volume of water required for the initial stimulation process.

This brief description illustrates that the combination of horizontal drilling and hydraulic fracturing has forged new connections at the water-energy nexus. These new connections warrant careful scrutiny as shale fuel extraction becomes more common globally, particularly because many large shale basins underlie arid regions. This paper provides a review and analysis of academic and non-academic scientific research concerning the nexus of shale fuel extraction and water resources. The purpose of this review is to summarize critical information for regulators, producers, and the general public while also highlighting key needs for future research. The paper proceeds in three sections: (1) a description of the literature collection process; (2) a summary of the literature in terms of data, methods, and findings; and (3) a discussion of the findings from our literature review, particularly focused on knowledge gaps and key unresolved debates. Our review is organized according to the shale fuel production process per Figure 2.2 below. We conclude with suggestions for future research.

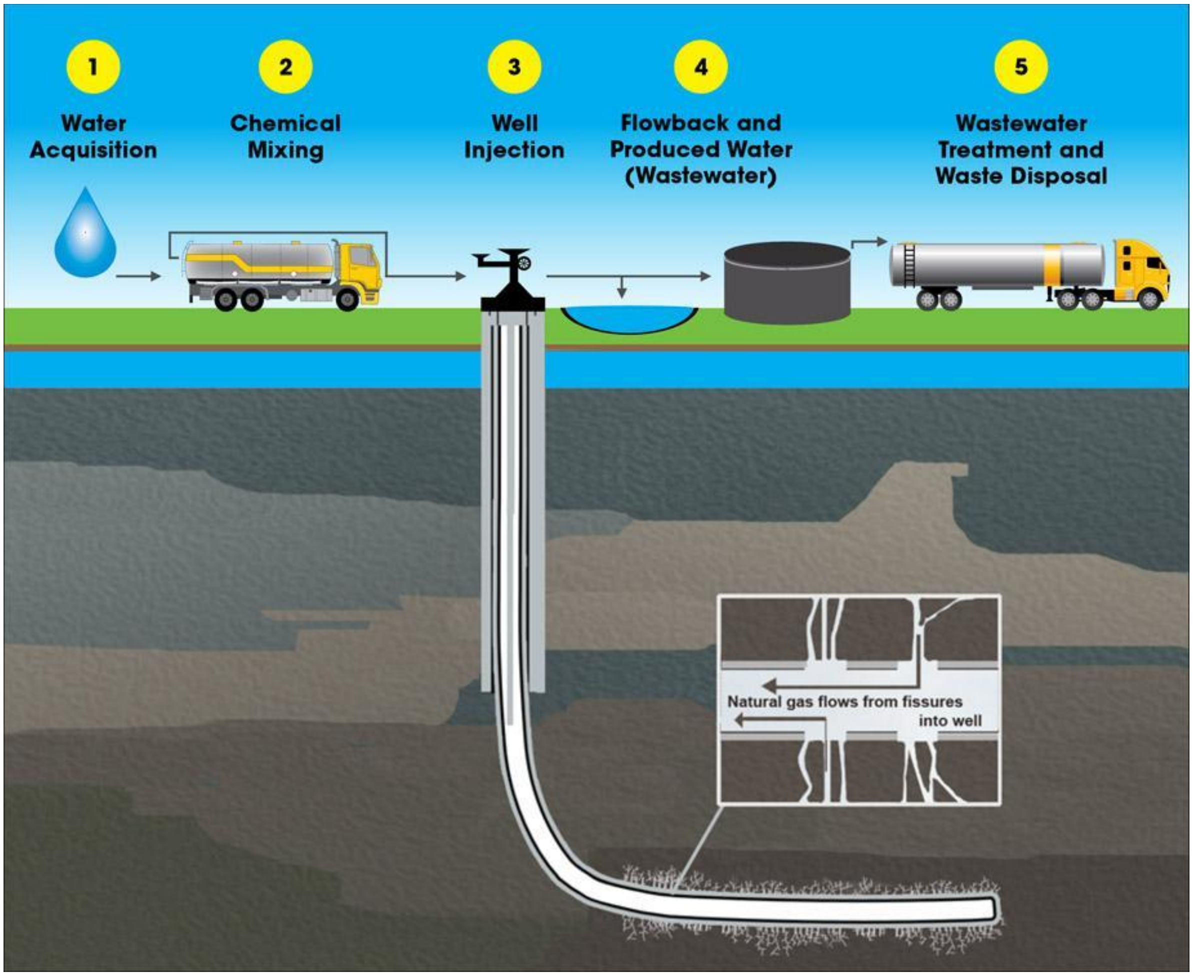

Figure 2.2 Depiction of the hydraulic fracturing process. Source: EPA 2015

Literature search: scope and selection process

To date, two systematic reviews of literature about hydraulic fracturing and water have been conducted (Lave and Lutz 2014; Rahm and Riha 2014). Lave and Lutz (2014) used a 'critical physical geography' lens to examine the biophysical and social dimensions of hydraulic fracturing. Water usage was considered in that review, but it was not the primary focus. Rahm and Riha (2014) focus specifically on water use and the risks to water resources from hydraulic fracturing. The authors conclude that current research on unconventional oil and gas is strongly biased toward conditions in the Marcellus Shale, and thus some results may not be easily applied to other regions. Building on these efforts, this paper considers published findings from a wide variety of disciplines and regions to further examine the impacts of hydraulic fracturing on water resources. We also consider non peer-reviewed publications from institutions such as the US Energy Information Agency (EIA), the United States Geological Survey (USGS), and US national laboratories.

Initial data collection for this review began with a keyword search using various combinations of "water," "shale," "water-energy nexus," "wastewater," "hydraulic fracturing," and "natural gas". Citations from the initial results were searched and cross-referenced to trace the research through a variety of disciplines including law, geography, and chemistry. This process was repeated monthly until May 2015 to keep abreast of new publications. As shown in Figure 2.3, scholarly interest in hydraulic fracturing has increased significantly since 2008, closely following growth in water usage with only a short lag and indicating the growing interest in the field. In total, 137 peer-reviewed publications were identified along with 12 non-peer reviewed publications from government organizations. The articles considered for this review also show nongovernmental research is clustered in several large, distinct clusters of researchers and institutions, together accounting for more than half the individual authors cited in this review. Of the 149 total sources, 110 report research findings related to the physical processes of water consumption, contamination, reuse, and wastewater disposal. The remaining articles focused on cultural or sociological impacts, air pollution, and other concepts not directly related to water quality or quantity, or to fuel extraction specifically. These articles tend to mention water as it relates to their subject matter—as an economic input, a possible driver of inequality, or simply for context—rather than as the primary focus of their articles. Where appropriate, we use articles that do not emphasize water resources for context only, as their conclusions are outside of the scope of this review. The review has a strong US bias, reflecting the fact that the majority of commercial-scale hydraulic fracturing activities, and research into such activities, are occurring in the US. Finally, to allow for ease of comparison, quantitative results from these studies are converted into common units.

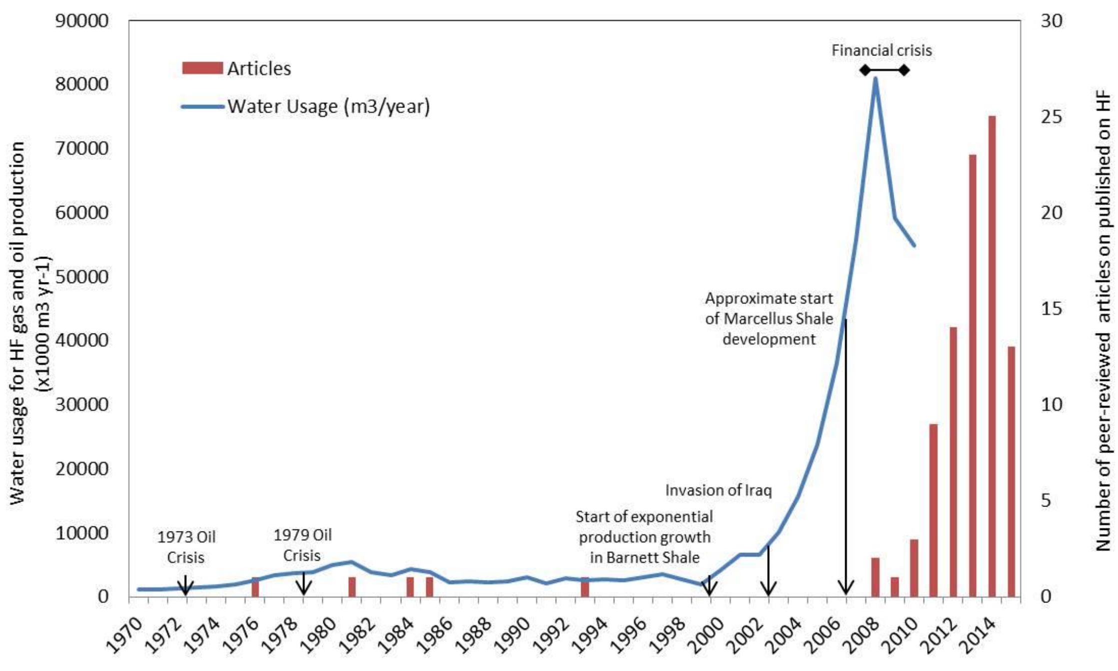

Figure 2.3 Graph showing the total yearly water usage in hydraulic fracturing from 1970-2010 compared with the number of peer-reviewed articles published on the subject from 1970-2015. Water usage data is from Gallegos and Varela (2015) and the peer-reviewed article data is from this review. Combined, the graphs show a clear relationship between the amount of water used and the interest of researchers in hydraulic fracturing.

Literature Review

Data Sources and Methods

A general summary of key sources of data and key methods can be found in Table 2.1. Nearly all the articles reviewed cite a small collection of regularly-updated governmental reports for background data (e.g. EIA Annual Energy Outlook 2011, 2012…) or consultancy reports. Governmental reports are open-access, and tend to rely heavily on peer-reviewed articles and data from outside organizations, though some reports do contain original data. The FracFocus Chemical Disclosure Registry is a major repository for primary data and partially falls under the governmental umbrella. Regulators in all states with major shale basins legally require disclosure of hydraulic fracturing data to FracFocus (FracFocus 2016),[4] as do many states with smaller shale industries. FracFocus is a free, publically-accessible database operated by the Groundwater Protection Council (GWPC) and the Interstate Oil and Gas Compact Commission (IOGCC) containing information on almost 100,000 shale wells. The information collected by FracFocus includes a variety of well identification data, lists of chemicals and their concentrations, and water volumes. FracFocus explicitly refrains from providing scientific analysis or arguments for or against hydraulic fracturing. In contrast, major consultancies such as SNL Energy and IHS advertise their analyses and decision support to industrial customers, with the goal of assisting them in strategy and operations. Access to their data and reports are available for a fee to any user, and many

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[4] Pennsylvania is in the process of creating a state-managed reporting database to eliminate its use of FracFocus. The system is expected to be operational in March 2016.

large institutions have agreements with these consultancies that allow cost-free access to individual users with appropriate credentials. It is important to note that the data organized by both FracFocus and these consultancies are self-reported by the industry (FracFocus; Nicot et al. 2014) , a collection method with an inherent risk of cheating, but which (when honestly reported) results in more accurate data.

Table 2.1: Sources of data found in literature

Data Source		Release Date	Description
Governmental Report (State or Federal)	DRAFT EPA Investigation of Groundwater Contamination near Pavillion, WY	2011	In-depth research on possible groundwater contamination near Pavillion, WY. Structured similarly to scholarly articles (background, methods, results and discussion, conclusion, references). Report released in draft in 2011, no official version released yet.
	USGS Data Series 868	2015	Temporally and spatially aggregated information on water use in hydraulic fracturing in the US since 1947. Data only; no analysis of environmental impacts. Does not discuss wastewater explicitly.
Grey Literature (Consulting Firms, Nonprofits)	SNL Energy "Summary of Shale Gas Wastewater Treatment and Disposal in Pennsylvania 2014"	ongoing	This consultancy focuses on industry-facing reporting and analysis of information pertaining to energy resources. Newsworthy items are distributed to subscribers several times each day, and thorough reports are collected and/or completed. This report uses publically-available data from PADEP to update a report from 2012 detailing shale gas wastewater quantities, chemical characteristics, and disposal methods.
	FracFocus	ongoing	This database provides a by-well list of chemicals and their concentrations used for hydraulic fracturing. Oil and gas companies are required by law to report this data, as well as identifying

			data (well location, date spudded and closed, etc.) in all major shale basins.

Broadly speaking, there are two common approaches to peer-reviewed research into the water-energy nexus as it relates to shale fuels: 1) a disaggregated approach, wherein a single step in the process is studied in great detail, usually empirically; and 2) lifecycle analysis (LCA), which aims to understand and quantify the flows of resources and materials throughout the entire production system, usually through modeling. A disaggregated approach can be time consuming and technically demanding, isolating one step from the rest of a process, but such an approach allows a thorough analysis of that step. Once the component steps of a process are understood, an LCA can combine these into a cohesive, “cradle to grave” story covering raw material extraction, production, use, end-of-life treatment, recycling, and final disposal (ISO 14040). However, because many disaggregated studies inform each LCA, LCAs must simplify inputs, and so create an idealized process (for instance, using mean or median values for each step), thus yielding idealized results. Results from LCAs will be discussed in section 3.2.5 below.

The state of knowledge

This section organizes results from the studies reviewed according to the stages involved in the process: drilling and cementing, fracturing, flowback/produced water, and wastewater management (see Figure 2.4). Where necessary, each section is further subdivided to focus on issues related to water quantity and water quality separately. Key findings are reviewed with emphasis on variations across different shale basins. In addition to the direct impacts on water resources, it is important to note that as many as 650 truck trips, 60-80% of all logistics movements for each well, are required to bring water and chemicals to each well pad and wastewater away from it (Rodriguez and Soeder 2015; Stark and Thompson 2013). The impacts of these logistical movements will not be discussed explicitly, but the movements themselves represent both a huge proportion of the cost for each pad and potential risk to the surrounding environment and society. Recognizing this, the industry has sought to minimize these costs by building temporary pipelines, less obtrusive and cheaper than so many trucks, to move water from streams and groundwater wells to well pads as well as storage tanks to hold flowback and produced water.

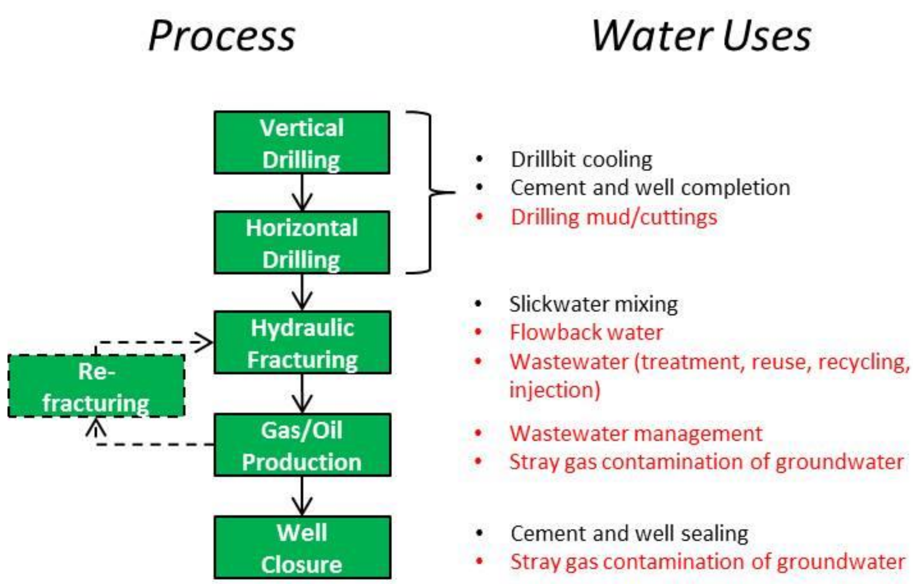


Figure 2.4 Conceptual model of the process of fuel extraction, modified to show the specifics of the hydraulic fracturing process. Important water considerations are shown next to the applicable phase, with wastewater considerations in red. Types of water usage that are common to all phases (i.e. equipment maintenance/cleaning) are omitted as their contribution to water use throughout the process is insignificant.

Drilling and Cementing

Water is used during the drilling process in order to create the drilling mud which keeps the drillbit cool, provide hydraulic pressure to drill the well, and then to mix the cement used to seal the wellbore. The vertical portion of the well typically uses less water than the horizontal portion(s). Taken together and in the context of the entire hydraulic fracturing process, the volumes of water required for these initial drilling steps are negligible; in the Marcellus region, for example, it accounts for approximately two per cent of a well's total use (Jiang, Hendrickson, and VanBriesen 2014). Water requirements for drilling are similar for conventional and unconventional wells, though conventional vertical wells require less drilling and therefore less water since they do not have laterals (Clark, Horner, and Harto 2013). Most of the observed variation in water use between shale plays at this stage is due to different geologies and technologies used, although an important variable is the length of laterals extending from the main vertical well. Lateral length varies on a well-to-well basis, even in individual basins and individual well pads, and is difficult to generalize (Scanlon, Reedy, and Nicot 2014). Dale et al. (2013) compared lateral length, drilling time, and fracturing water consumption in the Marcellus Shale across two time periods (2007-2010 and 2011-2012) and found that lateral lengths have increased over time, and that, although individual drilling companies showed more efficient water use, no trend was observed across the basin as a whole.

Hydraulic Fracturing

Water Quantity in Hydraulic Fracturing

Hydraulic fracturing is the most water-intensive stage in the construction of a shale fuel well, accounting for 85-95 percent of lifecycle water consumption (Jiang, Hendrickson, and VanBriesen 2014; Goodwin et al. 2014).[5] As shown in Figure 2.5 and Table 2.2, estimates of the volume of water injected during this process can vary substantially between and within shale plays. Most of the injected volume is freshwater, but a growing (though uncertain) volume is recycled flowback and/or produced water from earlier wells (see Table 2.3). The total volume of water required is largely dependent on two factors: technology and geology. Both can influence the number and length of stages in each lateral; in turn, both of these can vary widely. A stage is a section of the lateral that is fractured. Generally, the smaller each stage, the more fractures can be created and the more efficiently the resource can be extracted, but because the length of laterals varies, so does the number and length of stages. For example, shale wells in the Wattenberg Field in Colorado may have anywhere from 7-43 stages (median: 20; 10%-90% interval: 17-24), each averaging 76.2m in length (no standard deviation was given) (Goodwin et al. 2014). A different study, which did not specify the study basin(s), identified as many as 60 stages with lengths starting at 200m and gradually reducing to 50m as technology improved (Aguilera 2014). The downward trend in stage length over time illustrates one of the most important technical goals for the hydraulic fracturing

[5] **Consumptive water use:** water evaporated during production, lost underground, or embodied in a product; it results in a net loss of water in the watershed where the water originates and reduces the water availability of that region. **Non-consumptive water use:** denotes the water that is returned after use to the watershed where it originates; it may generate wastewater and result in degradation of water quality of the water region and/or increased costs to treat wastewater.

industry: the ability to fracture smaller stages to increase efficiency, thus using less water and less money (Kargbo, Wilhelm, and Campbell 2010).

Table 2.2: Estimates of the volume of water, in cubic meters, used to fracture one well in the Marcellus Shale.

Estimated Input Water per Well	
Source	**Estimate (cu m)**
Jiang et al 2014	3,500
USEPA (FracFocus 1.0)	4,136
NYSDEC	9,085
Rahm et al 2013	10,000
Lutz et al 2013	11,500
Rahm & Riha 2012	12,500
Brantley et al 2014	15,142
USEPA 2011	15,842
USEPA 2011	16,656
USEPA 2011	17,413
USEPA 2011	28,298
NYSDEC	29,526
Rahm et al 2013	30,000

Table 2.3: Estimates of the proportion of a new well's water input expected to come from recycled water. Some values repeat because of matching numbers in peer-reviewed literature.

% Recycled Flowback used in New Wells	
Source	**Estimate**
Rahm et al 2013	10%
Jiang et al 2014	10%
Mitchell et al 2013	12%
Jiang et al 2014	12%
Brantley et al 2014	13%
SRBC 2013	14%
Hansen et al 2013	18%
NETL 2013	25%

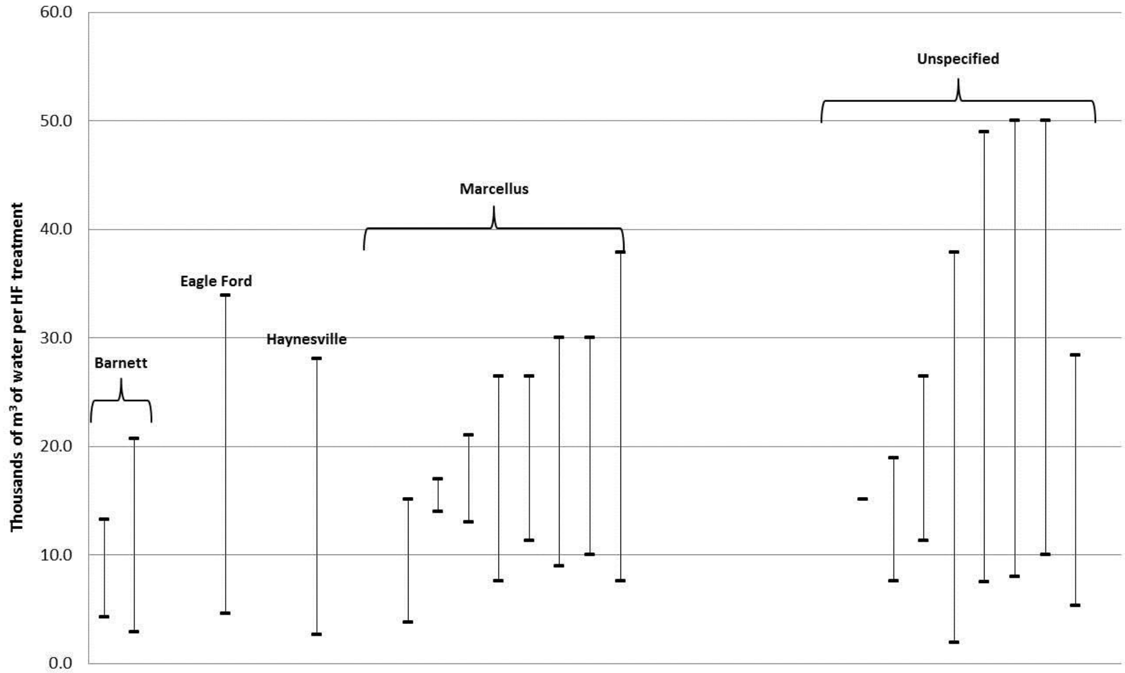


Figure 2.5 Visual representation of estimated volumes of water (m^3) used to fracture one well in the major US shale plays. The "unspecified" category refers to estimates that did not refer to any specific shale play. Each vertical bar is a value taken from a peer-reviewed publication. Data is from multiple peer-reviewed sources cited throughout this chapter.

Table 2.4: Selected results for water intensity for hydraulic fracturing operations on a per meter and/or per-unit energy basis. Note that many of these results have been converted into m^3 for ease of comparison.

Source	Basin	Qty of Water for HVHF
Brantley et al (2014)	Marcellus Shale	10.0 m^3/m
Murray (2013)	Arbuckle Shale	0.12 m^3/m
Murray (2013)	Woodford Shale	15.73 m^3/m
Goodwin et al (2014)	Wattenberg Shale	6.26-8.99 m^3/m .006-.01 m^3/MJ
Scanlon et al (2014)	Eagle Ford Shale	12.66 m^3/m
Scanlon et al (2014)	Bakken Shale	1.24-3.97 m^3/m
Jiang et al (2014)	Marcellus Shale	.0000094 m^3/MJ

Water Quality in Hydraulic Fracturing

A major concern of citizens living near hydraulic fracturing activities is the potential for natural gas and/or fracturing fluid to migrate into drinking water supplies.

Contamination of water resources could theoretically occur via three major pathways: (1) during drilling or by leakage through a borehole; (2) during hydraulic fracturing, by mixing with deep groundwater; and (3) surface spills or problems with treatment of the flowback and produced water that returns to the surface. Concerns were elevated after the 2010 film *Gasland*, which showed residents of Dimock, PA igniting tap water due to high methane concentrations. Despite the implied connection to increased hydraulic fracturing in the area, only one (much more recent) study has identified a likely direct connection between hydraulic fracturing and groundwater contamination (Llewellyn et al. 2015; discussed below). Most research into this subject tends to be inconclusive due to a combination of poor or nonexistent pre-drilling water quality data, the existence of possible sources of contamination other than hydraulic fracturing (for some pollutants), and substandard safety or quality control standards or poor execution of sufficient standards.

Several studies have suggested that a connection may exist between hydraulic fracturing operations and ground or surface water contamination in different regions, but most have indicated confounding variables or alternate causes that are at least as likely. One study, conducted in the Marcellus Shale, determined that waters from the deeper, briny aquifers (the sources of produced water) and the shallower, potable aquifers used for drinking water probably mix over very long time scales; however, because researchers could not find any indications of common fracturing fluid chemicals or hydrocarbons in the shallow aquifers, they surmised this mixing is probably due to natural fractures and other geologic processes, not induced by hydraulic fracturing

(Warner et al. 2012). Another study, undertaken in the Barnett Shale region of Texas, sampled 95 groundwater wells to determine if there was correlation between contaminant levels and proximity to gas wells. Their data showed a positive correlation between concentrations of arsenic, strontium, barium, and selenium—all associated with waste from natural gas extraction—and proximity to active gas wells. However, the same study showed that concentrations of methanol and ethanol, both common anti-corrosives in fracturing fluid, were not correlated with distance from nearby gas wells; further, almost half of the reference wells showed elevated concentrations of methanol and ethanol, both of which are naturally occurring (Fontenot et al. 2013). Finally, a 2011 draft report from the EPA indicated that contamination of the aquifer near Pavillion, WY with drilling cuttings was probably caused by hydraulic fracturing, but no peer-reviewed studies have been released on this subject, and the report itself remains in draft form (EPA 2011). Overall, the weight of evidence indicates that direct contamination of groundwater or surface water by hydraulic fracturing is not systemic (Vengosh et al. 2014). While there have been indications of water contamination near shale energy wells, this contamination cannot be conclusively linked to hydraulic fracturing operations, at least when proper safety and quality standards are maintained.

When safety and quality standards are not met, evidence is clearer. One example of this is methane migration into groundwater, and thus into potable aquifers and wells—the source of the infamous flammable tapwater. Many concerns about groundwater pollution by methane seem to be primarily due to wellbore leaks caused by poorly installed or decaying cement rather than upward migration of natural gas through newlyopened fractures (Olmstead et al. 2013). Contamination through this pathway is a common and well-studied occurrence in the petroleum industry and can even come from old, abandoned conventional gas wells, the majority of which have been neglected, sealed improperly or not at all (Vengosh et al. 2014; Brantley et al. 2014). The clearest evidence of water contamination directly attributable to hydraulic fracturing is found in a recent study by Llewellyn et al. (2015). The study compared the timing of nearby hydraulic fracturing operations and reported contamination of several groundwater wells to chemical analyses of those groundwater wells. The researchers found that water contamination was reported five months after hydraulic fracturing of a well that had previously been cited for contaminating a local spring with a fracturing fluid chemical, presenting strong circumstantial evidence of direct contamination. This circumstantial evidence aside, current scientific consensus suggests that water contamination during the hydraulic fracturing and well operation is almost certainly the result of poorly-sealed wellbores or large, unreported spills of fluid on the well pad rather than from the fracturing of subsurface rock.

It is also important to note that methane is not categorically considered a pollutant. Methane can be thermogenic, indicating it likely comes from a hydrocarbon source; biogenic, indicating it likely comes from a microbial source; or a mix of the two (Osborn et al. 2011). Given its prevalence in natural ecosystems, the presence of methane is not a conclusive indicator of contamination from hydraulic fracturing. It is particularly difficult to make this connection without historical, pre-drilling methane levels, which are usually unavailable.

Flowback and Produced Water

Quantity of Flowback and Produced Water

After each stage of a well has been fractured, pressure is released to allow the resource to flow to the surface. Every day over the first two weeks after all stages have been completed, hundreds or thousands of cubic meters of water returns to the surface. This is called flowback and it is composed mostly of the water that was injected for the fracturing treatment. The amount of injected water that returns to the surface has been shown to vary widely between 10 and 80 percent of injected volume (Vidic et al. 2013; Dale et al. 2013; Ferrar et al. 2013; Lutz, Lewis, and Doyle 2013; Maloney and Yoxtheimer 2012; Olmstead et al. 2013; Rahm and Riha 2014). Given such a wide variation, all 'average' rates or volumes of flowback assigned across a region conceals wide variation. Eventually the flow of water returning to the surface slows to 1-2 m^3 per day, most of which comes from the highly saline aquifers near the source rock; this is called produced water, and continues at this relatively low rate for the lifetime of the well. Flowback is estimated at approximately one third of the lifetime volume of water returning to the surface and the rest is produced water (Lutz et al 2013).

Quality of Flowback and Produced Water

Flowback water has similar chemical characteristics to injected fracturing water because, instead of a defined inflection point, there is a transition from flowback to produced water during which water quality changes from more like fracturing fluid to more like formation water (Barbot et al. 2013). This inflection point is difficult to pinpoint because longer contact time between underground fracturing fluid and the formation results in increasing dissolution of minerals from the formation, changing the relative proportions of fracturing fluid to formation water over time (Maguire-Boyle and Barron 2014, NETL 2009). The chemical signature of the initial stage of flowback is usually close to the composition of the fracturing fluid, but also contains dissolved elements from the formation. Chemical signatures can change from well to well because the precise combination of chemicals changes from well to well, but flowback generally contains some combination of antiscalants, anticoagulents, and other chemicals that are commonly found in fracturing fluid, in addition to very high total dissolved solids (TDS), sodium, chloride, naturally-occurring radioactive materials (NORMs), and barium from the formation (Abualfaraj, Gurian, and Olson 2014). Produced water tends to be very high in TDS and contains radioisotopes. Proper treatment has enormous importance—reused/recycled water must be sufficiently pure to allow maximum energy production; when it is returned to the environment, it must meet water quality standards. Understanding and conducting treatment properly, thus, is critical to the safety of the hydraulic fracturing process.

Contamination Risks and Wastewater Treatment

Wastewater management involves steps taken to reduce (the likelihood of) spills and leaks contaminating surface water and shallow groundwater as well as the accumulation of toxic compounds on soil (Vengosh et al. 2014). At most wells, flowback and produced waters are temporarily stored in lined pits or (increasingly) in tanks until entering a waste management system. The most common way to dispose of wastewater is through underground injection, whereby disused wells with suitable geology are used to store wastewater underground, up to 98% of the total wastewater volume in 2009 (Nicot et al. 2014; Rahm et al. 2013; Clark and Veil 2009). This practice has long been used in Texas and along the Gulf Coast for produced water from both conventional and unconventional wells (Nicot et al. 2014), where an estimated 50,000 injection wells exist (Vidic et al. 2013). Injection disposal is becoming increasingly common for flowback water from wells in the Marcellus Shale. However, because Pennsylvania geology is largely unsuited for injection wells, almost all water fated for injection must be transported to Ohio (Olmstead et al. 2013; Vidic et al. 2013). Injection wells are a relatively safe disposal method in terms of water pollution and have been used for decades, but in some areas (notably Ohio and Oklahoma) they can pose long-term and well-publicized seismic risks (both during the injection process and afterward) that are greater than those posed by the fracturing process itself (Vidic et al. 2013; Parker et al. 2014). In part because of the cost of transport, only about 27% of Marcellus Shale wastewater is transported to Ohio for injection, with the remainder of wastewater treated and/or reused in Pennsylvania.

Due to concerns about induced seismic activity and the costs of transporting wastewater long distances to injection sites, alternative management practices are being developed. Wastewater might be “reused”, meaning that it enters the hydraulic fracturing process at another location. Reused water is generally understood to require little treatment and may only need to be diluted. Produced water, which is usually high in salinity, is often used as a de-icer or dust suppressant, but this accounts for less than 1%

of the total wastewater volume (Rahm et al. 2013). Wastewater might also be “recycled”, meaning that it requires more involved treatment before use in other wells or for other industrial uses. This will be discussed in the following paragraph. The proportion of flowback/produced water that is reused or recycled has increased dramatically from 13 per cent to 90 per cent or more over the last decade (Brantley et al. 2014, NETL 2013). Reaching 100% reuse or recycling (of the flowback volume, not total injection volume) is probably not possible due to a gradual accumulation of organics and radioactive elements (Abualfaraj, Gurian, and Olson 2014). At present, on-site treatment before reuse or recycling is the most cost-effective because it does not require transportation, a major expense in hydraulic fracturing (Ferrar et al. 2013; Lester et al. 2015).

Water that is not injected, treated and reused, or spread for dust or ice abatement is shipped to wastewater treatment plants (WWTPs) and/or publically-owned treatment works (POTWs) before discharge to the environment (Wilson and VanBriesen 2012; Lutz, Lewis, and Doyle 2013). WWTPs tend to be designed with a specific type of wastewater involved—in this case, oil and gas wastewater (OGW)—but POTWs are generally designed for municipal wastewater, which has, among other things, a far lower concentration of total dissolved solids (TDS). During the early stages of Marcellus Shale development, POTWs were authorized by the Pennsylvania Department of Environmental Protection (PADEP) to treat oil and gas wastewater, provided it made up 1% or less of the total daily volume treated by each plant. Even at this level of dilution, the water quality downstream from these plants degraded rapidly, forcing PADEP to issue an advisory for over 300,000 Pennsylvanians downstream from treatment plants in the Monongahela River basin and spurring regulation in 2011 preventing the use of POTWs (Ferrar et al. 2013; Kargbo, Wilhelm, and Campbell 2010). In addition to these risks to and effects on human health, insufficiently treated wastewater can have major effects on downstream ecosystems because of its high concentrations of salts, metals, and NORMs liberated from the source formation (Vidic et al. 2013; Jiang, Hendrickson, and VanBriesen 2014). Although WWTPs are usually purpose-designed, concentrations of some chemicals in their effluent can still be thousands of times higher than upstream concentrations, and create a noticeable plume downstream, even after mixing with streamflow. This could impact ecosystems directly by damaging plants and wildlife that are sensitive to altered stream chemistry (Brittingham et al. 2014) or indirectly through bioaccumulation or groundwater contamination (Warner et al. 2013; Sang et al. 2014).

A final method of wastewater management is simply collecting and ignoring it. In parts of the Western US, it is common to leave flowback and produced water in open pits for the lifetime of the well, which can be 25 years or more. Over time though, evaporation of water from holding ponds leaves behind an increasingly concentrated solution that is more and more difficult to treat. The risk of contamination from these ponds leaching into the soil could even indicate a need for a Superfund-type cleanup at each well pad (Colborn et al. 2011).

Many technological solutions are being considered to improve the wastewater treatment system, a source of both substantial environmental risk and expense for the industry—drilling companies may spend much as $270,000 per well, about 15% of the total cost, on wastewater treatment (Jiang, Hendrickson, and VanBriesen 2014;

McGovern et al. 2014). One of the more intriguing ideas is blending acid/abandoned mine drainage (AMD) waters, a legacy of Pennsylvania's coal mining past, with flowback water in order to precipitate chemicals such as barium and radium (Kondash et al. 2014). This method reduces the concentrations of dissolved pollutants, but generates toxic solid waste that must be disposed. Laboratory-scale tests indicate this process is feasible, but it has yet to be tested at a field site. If large-scale tests are successful, this method would help mitigate two of the major risks to watersheds in Pennsylvania and anywhere else AMD and hydraulic fracturing flowback coexist. Guar gum, already a constituent of fracturing fluid, may also help to precipitate some solids out of wastewater (Lester et al. 2014). Shaffer et al. (2013) evaluated several different possibilities in terms of energy intensity, including mechanical vapor compression (MVC), membrane distillation, and forward osmosis. All are effective, though MVC requires a large supply of electricity (10 or more kWh per cubic meter of discharge) and the others show the best results with water that generally has lower TDS than typical flowback water. These solutions, though encouraging so far, have not yet been proven at the scale of an operational well.

Lifecycle Analyses

Lifecycle analyses serve to combine the findings from studies discussed above and present a holistic picture of the process. Findings are often reported as relational metrics such as liters of water per meter of lateral. One example comes from Clark, Horner, and Harto (2013) who use LCA to compare water use in four different shale plays (Barnett, Fayetteville, Haynesville, and Marcellus Shales). The study considers the entire lifecycle from well construction to final combustion in a vehicle or an electrical power plant. Findings are reported as a water intensity per unit of energy production (i.e., L/GJ). The authors determined that conventional unassociated gas is the least water-intensive (about 1 L/GJ) among all fossil fuel extraction processes with hydraulic fracturing ranging from about 1.25-2 L/GJ depending on the shale play. Dale et al. (2013) study a wider range of costs and benefits, including greenhouse gas (GHG) emissions, water use, energy consumption, and energy return on investment (EROI). This study compared Marcellus Shale well construction from 2007-2010 to well construction from 2011-2012. The authors show that average GHG emissions are nearly a quarter lower than they had been, and energy consumption and water use have also fallen, though by much smaller proportions; in terms of water consumption, improvements in the fracturing process have likely been offset by the increasing length of each well's laterals.

Discussion of Literature

The shale fuel extraction industry, though decades in the making, has only recently attained the scale to reconfigure the water-energy nexus in profound ways. By combining detailed empirical work at each step of the process with comprehensive life cycle analyses, knowledge of the process as a whole is vastly improved. However, gaps still remain and we face the challenge of disseminating the knowledge garnered in existing shale plays and applying it to potential plays before drilling becomes more intense. A critical assessment of the literature above yields three main insights that should underpin research moving forward: relational terms are enormously useful for comparing changes over time and should be used as often as possible; wastewater is well-understood, but the larger-scale implications of that wastewater is not; and finally, research on hydraulic fracturing would benefit from explicit focus on region and scale to better quantify and understand local impacts. The following sections will discuss each of these in turn.

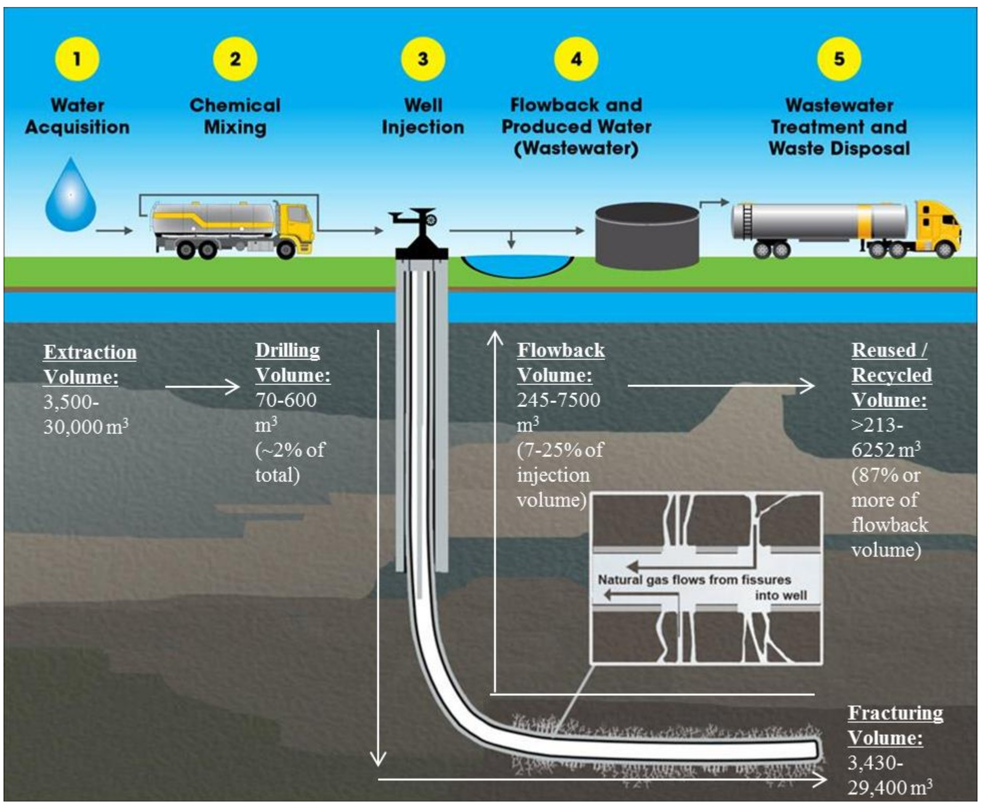


Figure 2.6 Ranges of water use broken down by stage, taken from literature reviewed in this chapter. Note that flowback volume is only what returns over the first two weeks or so of production. This figure does not show lifetime produced water volumes and their reuse/recycling fates. Figure is modified from EPA (2015).

Reporting Results

Overall results from the literature are presented in Figure 2.6 as ranges. Aside from lifecycle analyses, very few studies report water use values in relational metrics, though there are notable exceptions (e.g., Scanlon, Reedy, and Nicot, 2014). Results are most commonly reported in terms of total volume per well (L) or in total water use across a region or shale play. Although these absolute volumes are important, basin-level or technology-level comparisons are difficult or impossible without some common, relational denominator such as volume per lateral length (e.g., L/m), volume relative to local water availability (L used for hydraulic fracturing/L available in the environment), or volume per unit energy produced (e.g., L/GJ). Reporting water use in these relational terms will enable analysts and decision-makers to project total water use against water availability, to better understand patterns and trends in water-use efficiency as technologies improve by well production declines, to compare various forms of energy production, and to assess the future prospects of hydraulic fracturing in regions expected to be impacted by climate change.

Wastewater and Wastewater Management

Wastewater management might be the most widely-researched aspect of shale energy production: its chemical composition is generally well-understood, and numerous ideas for treating or neutralizing it have been proposed. All that is lacking is an understanding of how much of it is produced, and whether that number is important.

There is a wide range of values for the proportion of flowback relative to the volume injected (Brantley et al. 2014; Vidic et al. 2013; Shaffer et al. 2013). The amount of water that returns to the surface as flowback in the Marcellus Shale (that is, in the first two weeks or so after the well is put into production) generally falls between 10% and 80% of the injected volume, with an average of 10%. Literature also generally agrees that the vast majority (87% or more) of flowback is reused or recycled. Unfortunately, no data exists on where or when that water is reused. Any water transported out of the watershed in which it originated, whether for reuse or treatment, represents an absolute loss to the originating watershed with unknown ecosystem impacts.

Further, reused/recycled water is usually presented in relation to injected volume. This can result in misleading "headline" numbers: reusing 87% of flowback (the headline number from Brantley et al. (2014)) is admirable under most circumstances, but if an average of 10% of the injected volume returns as flowback then only 8.7% of the initial (freshwater) injection volume is actually reused. This net balance has much different implications for local ecosystems than the headline 87 percent (Rodriguez and Soeder 2015; Hansen et al 2013). The rest of the water remains trapped underground to return throughout the lifetime of the well, if at all—there is little to no evidence showing that fracturing fluid and groundwater, separated by thousands of vertical feet, mix, so at least some of the water from each shale well is effectively permanently removed from the hydrological cycle. What does return, if it is treated and released to the same watershed from which it was extracted (not transported across watershed boundaries for reuse or treatment), may not come back for years or decades. Again, this would have unknown ecosystem impacts: consumption of a few thousand cubic meters of water may be negligible on the state or national level and on a long time scale, but in an individual watershed in the short term, it could be significant and even catastrophic.

Despite exponentially growing scholarship (Figure 2.4), there is still uncertainty about the ways in which flowback water impacts water quality. Wastewater quality research has been slowed by two factors common to all studies: a lack of baseline water data (sometimes coupled with environmental "noise" such as naturally-produced contaminants and pollution from historical coal or oil extraction) (Brantley et al. 2014; Vidic et al. 2013; Vengosh et al. 2014) and the exemption granted to unconventional oil and gas companies allowing them to keep proprietary fracturing fluid mixtures secret, as long as they do not contain diesel fuel (the so-called "Halliburton Loophole" to the federal Energy Policy Act of 2005). The former makes it difficult to determine pre-extraction conditions and, therefore, to attribute observed impacts specifically to hydraulic fracturing even where correlations in space and time are identified. The latter has made quality analysis and modeling of water injected for hydraulic fracturing difficult due to lack of sufficient information about inputs, except where companies have voluntarily disclosed the concentrations of chemicals present in the injectate. Further, the poor understanding of the composition of fracturing fluid means that it is difficult to project the environmental impacts on downstream (or subsurface) ecosystems and water resources. This also makes it difficult to design industrial wastewater treatment plants optimized to efficiently clean flowback and produced waters to a level where they can be safely returned to the environment.

The lack of baseline data is being slowly remediated as state agencies and researchers begin to establish long-term monitoring programs (Rhodes and Horton 2015), but these data will not be available for years, and in any case will already include the impacts of the initial phases of hydraulic fracturing. Further, not all contaminants or indicators of hydraulic fracturing activity are regulated: methane, for instance, is not necessarily considered a pollutant, and the Department of the Interior does not recommend any remedial actions unless concentrations are sufficient to make it an explosive hazard and cause precipitation of solid minerals from water (Vidic et al. 2013).

The final water consumption of shale energy has the potential to fall significantly with increased reuse of flowback and produced waters for fracturing and/or use of alternate fracturing technologies such as propane gel, liquid carbon dioxide, and nitrogen foam (Rodriguez and Soeder 2015). However, water use per unit energy production should be viewed with more caution than water use per unit lateral length measures. With the exception of re-stimulation, water use and lateral length do not change after the initial treatment, but estimated ultimate recoveries (EURs) for shale energy wells are continually refined over the lifetime of the well (Ikonnikova et al. 2015), thus changing the water use per unit energy value.

Scale and Water Sources

Clearly, spatial and temporal scale is important when discussing water consumption. Researchers often generalize the implications of water use, making the accurate but unquantifiable point that water usage can be damaging even in relatively water-rich regions depending on the volume extracted, the duration of extraction, and the time of year (Hammer, VanBriesen, and Levine 2012). Water consumption for hydraulic fracturing is commonly reported at the state level, and often expressed as a proportion of total water consumption. At that scale, consumption seems negligible, concealing substantial spatial and temporal variation at the county and watershed level. No studies included in this review conducted careful spatio-temporal analyses to provide details about where and at what time of year particular watersheds might be vulnerable to hydraulic fracturing activities. Such local assessments must be undertaken to determine where and when water extraction for energy production can occur without compromising human or ecosystem water resources. Such assessments should determine the quantity of water available for safe extraction (Hoekstra et al 2011) with careful limits based on locational and seasonal considerations (Rahm and Riha 2014).

The implications of groundwater availability are further reason to conduct location-based analyses. Knowing when and where groundwater is required will vastly improve our understanding of the water sustainability of hydraulic fracturing. In other words, future research into the risks of water extraction must work toward a geographically explicit approach, which considers surface water – groundwater balances.

Research into the water use and impacts of hydraulic fracturing has been conducted at multiple geographic scales, which presents monitoring and measuring problems that have already been acknowledged (Rahm and Riha 2012). This is not a problem unique to the research on hydraulic fracturing (Hussey and Pittock 2012; Bazilian et al. 2011), and scholarship surrounding hydraulic fracturing would be well-served by an agreement to measure water use and energy production on geological or hydrological scales (such as the shale basin for energy or the watershed for water use) rather than administrative scales (county, state, or country) which often overlap or are overlapped by multiple physical scales. Information presented in this fashion would be better able to capture physical similarities between basins, though it could admittedly conceal possible impacts of differing legal regimes.

Conclusion

The long-term water balance of shale fuel production is not yet clear because most wells have yet to reach the end of their productive lifetimes (Rahm et al. 2013). Further detailed analysis of water throughout the process of shale energy extraction is needed to complement existing lifecycle analyses. This research must be exhaustive in terms of documenting and understanding where input water comes from, how much of it is used, how much water returns from wells, and how and where it is treated or disposed. Future studies should also present results in comparable terms, including water consumption per meter of lateral or per unit of energy produced. Further, increasingly small-scale research should be encouraged to determine basin-by-basin and watershed-by-watershed impacts on environment and society. All of these details matter in order to move toward comprehensive understanding of water use in hydraulic fracturing, which is important as hydraulic fracturing becomes more common and adds stress to previously unaffected watersheds. By understanding these factors, policymakers will be better-informed to make decisions that can help to maximize available energy resources, ensure public health and safety, and create a sustainable energy extraction and production system.

Chapter 3 Concepts and Methodology

Conceptual Framework: Political-Industrial Ecology

Political ecology and industrial ecology are two major theoretical and conceptual frameworks, thus far largely separate, by which we might come to better understand and therefore resolve the complex, multi-scalar, and multi-disciplinary issues of human-environment interactions. The primary focus of political ecology is describing how political and environmental outcomes are linked and shaped by sociopolitical factors (Cousins and Newell 2015). Industrial ecology focuses on describing and optimizing the flows of material and energy through a particular industrial system or industrial ecosystem (Erkman 1997). The purpose of this section is to argue that the combination of these conceptual frameworks can lead to a more comprehensive understanding of human-created systems and their interactions with the environment and society. The result of this combination, political-industrial ecology, is a recent and growing hybridization of the legal, societal, and environmental traditions of political ecology and the process-based, technology-oriented aspects of industrial ecology. Political ecology provides tools well-suited to deconstructing the false (though inadvertent and arguably necessary) indifference to societal influences presented by much of the process-based research into HVHF. Industrial ecology's history suggests it can provide solutions that optimize the costs of HVHF, monetary or otherwise, and provide actionable recommendations for "better fracking," however defined. HVHF is an industrial process with a profound impact on many areas of society in general and the water-energy nexus in particular; parallel exploration of society, industry, and environment provides the fullest understanding possible. As such, political-industrial ecology provides a powerful lens by which to interpret the ways in which HVHF reconfigures the water-energy nexus.

In support of this assertion, this chapter will first separately explore selected political and industrial ecology literatures. Discussing the strengths and weaknesses of each framework will illustrate cracks in each that would be well-reinforced by the other. In order to further construct the foundation for political-industrial ecology, I will next contend that political-industrial ecology ideas have existed for decades, though it is most often relegated to the role of narrative underpinning in political ecology or found as offhanded remarks in the discussion sections of industrial ecology articles. This history powerfully advances the argument that neither political ecology nor industrial ecology can exist in a vacuum. Finally, I will suggest that HVHF is a political-industrial ecology and should be analyzed as such, creating the conceptual scaffolding on which the remainder of the paper will build.

Political Ecology

Political ecology is most succinctly explained as a "broadly defined political economy," as it interacts with ecology (Blaikie and Brookfield 1987; 17), though a number of increasingly specific definitions have been presented. Political ecology expands on political economy's focus on politics, law, and the economy by adding a focus on ecology—the relationship between organisms and their environment, as well as the characteristics of both of those. The result, a “fusing of biogeophysical processes with broadly social ones” (Zimmerer 2000), attempts to describe the relationship between a society shaped by its environment and an environment shaped by the society that inhabits it. The roots of political ecology as a whole trace to a wide variety of fields and conceptual understandings, including cultural ecology, hazards research, feminist development studies, Marxist political economy, peasant studies, and many more. These influences have collected around five primary political ecological traditions: environmental subjects and identities, conservation and control, environmental conflict, degradation and marginalization, and political actors and objects (Robbins 2012). Though any of these traditions could be applied to study of hydraulic fracturing, this section will focus on the influences from the last two.

The tradition of degradation and marginalization in political ecology states that development efforts to improve production systems of local people may lead to decreased sustainability and, paradoxically, increased poverty and inequality (Robbins 2012). This idea stems from the Marxist economic idea that capitalism inevitably degrades the environment as it seeks to extract surpluses from it. Wealthier regions tend to outsource this degradation to poorer ones, benefitting from the productive capacities of those regions while simultaneously placing on those areas the risks of environmental degradation and/or social marginalization. As these areas become degraded, they become less economically valuable and thus increasingly politically and socially marginal. The linked ideas of degradation and marginalization show clear influence from classical economics, specifically the assumption of a rational producer. Classical economics simply assumes rationality—for example, the producer will attempt to expand production to maximize profits. Marxist economics views degradation and marginalization as an unavoidable consequence of capitalism and class struggle. Social science, however, is rarely as simple as either framework suggests. Recognizing this, political ecology analyzes the "chains of explanation" that shape this rationality at scales ranging from the individual producer to globe-spanning trade systems in ways that classical and Marxist economics do not.

A more recent response has emerged in opposition to the Marxist hypothesis of inevitable destruction: ecological modernization. Ecological modernization is the idea that methods of production, one of which is nature, can be harnessed to reverse the inevitable destruction of the environment (Buttel 2000). It arose in response to the widely-accepted notion of sustainable development, which was primarily focused on rural areas and the Global South. It takes a capitalist approach to degradation and marginalization: rather than tracing the evils of the capitalist system to their (inevitable?) conclusion, ecological modernization provides a lens for understanding problems and solutions as they apply to the advanced industrialized world, thereby lessening (or ideally, eliminating) the risks of degradation and marginalization. By viewing nature as an element of industrial productive force and ensuring it is sustained rather than destroyed (Spaargaren and Mol 1992), ecological modernization seeks to change contemporary society rather than dismantle it. Though ecological modernization was started as a sociological way of understanding society, it has spread to other social sciences, including geography, resulting in critiques from some practitioners. Many of these critiques have centered around environmental (in)justice for example, solar energy, an industry emblematic of ecological modernization, but that also has far-reaching commodity and supply chain effects (Mulvaney 2013). A review of such political ecological critiques however indicates that what political ecology has been done inside the industrial workplace has primarily focused on occupational hazards (the "industrial hazard regime"), and has done little to illuminate the industrial black box (Barca and Bridge 2015).

Geographers have also added an interest in "proper" scale to the framework (K. S. Zimmerer and Bassett 2003b, p 288). A geographic approach to ecological modernization rejects the assumption that scale is not a vertical series of politically-established boxes (global, national, state, etc.) but a multifaceted concept that must include "non-traditional" scales, particularly ecological ones, which are not typically studied. Ecological modernization, particularly by focusing on ecological scales, helps keep the ecology in political ecology to paraphrase Walker (2005), a long-held but not always achieved goal of political ecology (K. S. Zimmerer and Bassett 2003, p 1). The degradation and marginalization tradition of political ecology, which implies a unidirectional relationship between humanity and the environment, subordinates ecology to a mere stage upon which the great struggles of humanity are played. One cannot, however, ignore ecology because its characteristics have a major influence on the way in which we interact with the environment and thus the way our society is structured. The influence of environmental characteristics is the foundation of political ecology's tradition of political actors and objects.

This tradition holds that the "material characteristics of non-human nature impinge upon the world of human struggles and are intertwined with them, and so are inevitably political. Yet, as these characteristics and agents assume new roles and take on new importance, they are also transformed by these interactions" (Robbins 2012; 23). Political actors and objects provide a way of understanding how the nonhuman characteristics of natural things influence the politics of humanity, thus infusing those natural things with political influence. Consider shale oil as a natural resource. Its inherent characteristics would exist regardless of whether it was extracted or not. Using hydraulic fracturing, these non-human characteristics are brought to the literal and metaphorical surface and impinge upon and become intertwined with humanity, becoming inevitably political—"natural resources are not naturally resources, but they become so because of cultural, economic, and political factors" (Bridge 2011). The material characteristics of the shale oil are transformed into power, both the energetic sort and the geopolitical sort, and societal systems are created (hydraulic fracturing), reinforced (hydrocarbon dependency), and transformed (water governance) by the newly-important (non-human) characteristics of the oil. The ecology of the oil—its energy density, its geographic location, and so on—becomes political, and in turn shapes politics.: the politics and the ecology are equally important.

While the political ecology traditions of degradation and marginalization and political objects and actors describe complex chains of explanation, they do not paint a full picture of humanity's interaction with nature. Rather, they focus on the causes and effects of interactions between humanity and nature. These causes and effects are important, but equally important are the inner workings of the processes at the socio-nature interface. These processes are known as industrial systems. As in political ecology, industrial processes are shaped by society and ecology. Industrial processes are human-created transformative processes that are shaped by the environment, social and ecological, in which they operate. The technologies they use are closely governed by both human ideas—the need to change a raw material into a finished product—and the inherent, non-human characteristics of the raw material at issue—a shale oil well is a much different system than a lumber mill. The study of these systems and their interactions with each other and the environment is known as industrial ecology.

Industrial Ecology

Industrial ecology seeks to explain interactions between industry and the environment using the industrial process as the primary focus. Operationalizing sustainable development is an important piece, and industrial ecology research has the explicit goal of creating policy-relevant solutions for engineering and cost-optimization problems that allow maximization of available inputs and outputs (including "waste" products) and therefore profits (Ayres 2000; Erkman 1997; Graedel 1996). Industrial ecology has sometimes used a too-broad definition of industry: that which "encompass[es] the sum total of human activity…the impacts on the planet resulting from the presence and actions of humans…[and] society's use of resources of all kinds" (Graedel 1996). However, it is difficult to argue that anything a human does is industrial. There are innumerable examples of things humans and animals both do in practically the same way, for example. Such a definition also implies humans are wholly separate from nature (Graedel 1996), falsely describing the human-environment relationship as one-directional. Here I propose an alternate definition of industry: the intentional, humandirected transformation of a raw material, idea, existing good, or combination of these into a new good or idea through a specific, predictable (but adjustable) process. This definition allows one to imagine an individual industrial process, an industrial system of many linked industrial processes, and envision both as influenced by and influencing natural ecosystems.

Industrial ecology takes inspiration from ecology in two ways: imagining individual processes as "organisms" and industrial systems (or the industrial system as a whole) as "ecosystems." Individual industrial systems transform inputs into products and wastes, operating in a way analogous to an organism's metabolism: industrial "organisms" consume raw materials or the products of other industrial processes (such as water, sand, steel, etc.),[6] digest them through a specific, predictable process (HVHF), and produce a predictable output (shale oil and/or gas). This metabolic metaphor is a powerful way to interpret industrial systems.

A set of industrial processes linked together may be considered analogous to the food web. A natural ecosystem (broadly) consists of a series of organisms and biological processes that feed on each other—plants absorb sunlight and soil nutrients to grow, are eaten by herbivores who are eaten by carnivores who die and are turned into soil nutrients; all is consumed in this system, so there is no waste when the process is viewed as a whole. A similar process happens in an ideal industrial ecosystem: the products and wastes from one organism/process are used as inputs for others, with the goal of reducing

[6] Raw materials are known as "elementary flows" to industrial ecologists, and are defined as "material or energy entering the system in question that has been drawn from the environment without previous human transformation; or material or energy leaving the system in question that is released into the environment without subsequent human transformation."

or eliminating waste (Erkman 1997). In practice, this is probably impossible (Erkman 1997), but it represents an aspirational goal for industrial ecologists, whose long-term goal is to move humanity forward to an industrial system balanced (sustainable) in terms of resource use (energy inputs are not included; effectively unlimited energy is assumed to come from the sun) (Graedel 1996). This conceptualization of industry creates an ecology separate from the natural ecosystem, but closely linked because the industrial ecosystem relies on the natural one for resources and services. Industrial ecology thus recognizes that industrial systems cannot be viewed in isolation from surrounding systems (Ayres 1995).

It is important to note here that both products and wastes should be considered in an industrial ecology, as would be the case in natural ecology (Frosch 1992; Frosch 1996). A long tradition of industrial ecology has focused on system optimization through waste minimization, as would happen in a natural ecosystem (Erkman 1997). However, such “end-of-pipe” studies are not the only form of industrial ecology, nor are they always a desirable goal for the industrial ecologist. For example, some forms of “waste,” which might be useful to other firms in the industrial web, might be most useful to the system if they were maximized instead of minimized.

Industrial ecology methods generally involve recommending policy solutions based off of measured differences in the harm caused by various products or processes in order to optimize efficiency or costs (Ayres 1995). However, comparing two different products of processes is often impossible—how does one choose between a non-biodegradable product manufactured sustainably and a biodegradable product manufactured unsustainably? Industrial ecology’s most common method, the lifecycle analysis, answers this question by measuring everything in terms of energy use throughout the lifecycle of a product. This is informed by the belief that, in theory at least, a product that requires less energy throughout its life is more sustainable and therefore a better choice. Such thinking gave rise to the idea of "footprints" (i.e. "carbon footprint," "water footprint," etc.). This industrial thinking, by emphasizing the objectivity presented by comparing numbers, has enormous societal power and influence.

A combined political-industrial understanding of the world increases the conceptual power of each framework, as each complements the other's weaknesses. With a stated goal of "optimizing" industrial systems, and the veneer of mathematical precision and objectivity that implies, industrial ecologists obscure the many political assumptions and political implications inherent in their work. Deconstructing those assumptions is required—what is the goal of this optimization? Why is that the goal? For whom is the solution optimal? The industrial ecologist also ignores the impact the process itself has on the surrounding society—will this process change the society that created it? If so, how? Unpacking these questions is the bailiwick of political ecology, and industrial ecology would benefit greatly from such ways of thinking. At the same time, both ecologies share an interest in policy-relevant questions and solutions using different perspectives. Political ecology identifies distant and/or ambiguous chains of explanation, but in many cases changing the processes involved, optimizing the arrangement of technology, or recommending new technology altogether can solve the problems identified (in part at least). That industrial ecology's concepts have enormous social and political acceptance helps.

In sum, political ecology uncovers problems causing and caused by political, economic, or social inequality; industrial ecology applies and/or rearranges processes and technology to solve problems. Neither is false; both are incomplete. Industrial ecology requires political ecology's descriptions of chains of explanation, and political ecology needs industrial ecology's technology and process focus. The result, political-industrial ecology, is a powerful, ethically-informed, policy-relevant way of thinking that holds great potential for improving the human condition in all locations and at all scales.

Toward a political-industrial ecology

To an extent, there has always been conceptual frameworks and analytical practices that might be representative of political-industrial ecology, but operating without the name. For decades, industrial organizations have been recognized as social systems, not mechanical ones (Barnard 1963). Establishing a social system such as a firm simultaneously creates a community with its own culture and norms, negotiated through interactions between individuals and groups of varying power and influence. Put another way, a new environmental action (establishment of an industrial system) spurred the emergence of new environmental subjects (employees, among others) who are further transformed through their interactions with existing political actors and non-human political objects. Later authors acknowledged that industrial ecology would require "rethink[ing] how we want to regulate waste materials of all kinds" (Frosch 1992) and that technological improvement (which industrial ecologists see as inevitable) will be "…faster if it is stimulated by scarcities, carbon taxes, or regulatory constraints…"

(Ayres 2000). Both statements, as well as industrial systems themselves, are replete with political implications that are largely unexamined, a blind spot acknowledged by some in the political ecology community (Barca and Bridge 2015). Finally, despite much focus on monetary measures of cost-optimization and profit-maximization, at least some industrial ecologists acknowledge that there are limitations inherent in quantifying all forms of value in monetary terms (Ayres 1995). Such acknowledgement provides an excellent starting point to combine political and industrial ecology to understand experiences of degradation and marginalization and the different value systems that surround industrial development, which are often implicit rather than economic.

Political-industrial ecology literature (or rather literature that is consciously political-industrial) is relatively sparse. Industrial ecology research acknowledges that societal processes are important, but rarely studies them in a meaningful way; political ecologists often use industrial processes as a framing device, but rarely are the workings of the technology analyzed for modification. Interestingly, much of the research has focused on resources, especially energy and water. Biofuels, like HVHF a major reshaping of the water-energy nexus, play an important role in the former. Discourses that construct “wastelands” (Baka 2013; Baka 2014) as ideal places to grow biofuel crops for conversion into liquid transport fuel are called into question, especially when previous research had found that the energy produced by existing biofuel crops is far greater (Baka 2012). Such a research project shows obvious influences from political ecology (marginal lands) and industrial ecology (measuring energy output), and exposes contradictions. A different set of papers views the water supply system for Los Angeles through a political-industrial lens (Newell and Cousins 2014; Cousins and Newell 2015).

These papers view Los Angeles as an urban organism, using the urban metabolism metaphor. They identify the Los Angeles Aqueduct as an industrial system with enormous social influence throughout as the impact of Los Angeles' water consumption is outsourced to relatively powerless Owens Valley. Once problems arose, Angelinos paid for the water they took, but the power imbalance remained. This imbalance was created by the industrial system of an aqueduct, which then solidified the imbalance through sheer size and inertia.

Hydraulic fracturing has all of the markings of a political-industrial ecology. The system itself is well-understood in a step-by-step manner, as detailed in Chapter 2 of this thesis. One well can be considered the "organism" in the ecological metaphor, metabolizing large sums of water in order to yield fuel. Hydraulic fracturing is also an industrial ecosystem: many wells (organisms) certainly, but each well requires inputs from other industrial processes (steel, transport, even other oil and gas operations for energy) in addition to the elementary flows (water, sand, etc.). These inputs are closely linked with their societal valuation as well as their material characteristics: enormous volumes of water are used for HVHF because water can carry the chemicals used to strip hydrocarbons from shale, but it is also far more available (and cheaper) than existing alternatives such as carbon dioxide. Water is an industrial input to HVHF, but it also exposes sociopolitical questions, many of which revolve around water use: if water used for HVHF is consumed entirely, what are the impacts on farmers, tourists, and other groups whose environmental identity is represented in that water?

A HVHF well also creates outputs that feed back into the industrial ecosystem—most obviously oil and/or gas, but also wastewater. The uses of oil and gas are many, but collectively they reinforce one major societal norm: reliance on hydrocarbon-based energy. That reliance has done much to shape the global economy. On a smaller scale, wastewater, as a waste product, might be considered extraneous, but it also has regional impacts on ecology (polluted rivers) and governance (restriction of untreated discharge into rivers). Of course, people work at wastewater treatment plants, which rely on construction workers to build them, creating a political-industrial ecology dependent on and reinforcing the existing one surrounding hydraulic fracturing.

Political-industrial ecology combines industrial ecology's process-based focus with the societal influences of political ecology, while ensuring ecology and the environment remain important rather than being considered context. Such a framework is ideal for hydraulic fracturing. The industrial process that is hydraulic fracturing is part of a web of industries (steel, hydrocarbon, construction, and more) all of which are governed in various ways. HVHF also requires ecological inputs (mostly sand and water), also governed. The creation of the process is a reflection of society's interest in maintaining a hydrocarbon-based economy, and the process' continued existence reinforces that economy. Water use is a highly effective entry point for a political-industrial ecology: it is governed by a number of organizations at different scales and it is the primary ecological input to the industrial process, without which the process could not exist. The following chapter will discuss the ways and scales which I will use to analyze this political-industrial system.

Chapter 4 Methods

Research Questions

Given the water demands of HVHF and the increasing rate at which it is being deployed in the US and abroad, the purpose of this research is to answer the following questions: first, what is the sustainable extraction volume in each USGS cataloging unit[7] in Pennsylvania, accounting for seasonal discharge variations; second, which Pennsylvania cataloging units, if any, are at risk due to increased HVHF under high, medium, and low water use and intensity scenarios; third, what regulatory frameworks exist regarding water extraction, and are they sufficient to protect watersheds from over-extraction for hydraulic fracturing? Consistent with methodologies in political industrial ecology (e.g., see Cousins and Newell, 20105), these questions will be answered using a combination of spreadsheet-based modeling and content analysis of existing regulations related to water management.

Scale of analysis

Research into the water use for and impacts of hydraulic fracturing is challenged by two primary scalar issues: the modifiable areal unit problem and the mismatch

[7]Cataloging Unit: the smallest commonly-used partition in the hierarchy of hydrologic units. It is a geographic area representing part or all of a surface drainage basin, a combination of drainage basins, or a distinct hydrologic feature. These units subdivide hydrologic subregions and accounting units into smaller areas. There are 2264 cataloging units in the US. Cataloging units may be colloquially referred to as "watersheds," but that is a non-specific term that can be applied to a basin of any size. (USGS.gov)

between the measuring unit and the hydrological unit. In its general formulation, the modifiable areal unit problem (MAUP) refers to the fact that the results of a study can differ simply as a function of how boundaries are drawn and how data are aggregated (Openshaw 1984; Dark and Bram 2007). Hydraulic fracturing research, most often conducted at the national, state, and well-pad scale, is often affected by this problem. Analyses of water use presented at the scale of an individual well might sound significant and be interpreted as such; but when aggregated over a large area, the quantity of water may be small compared to other uses of water. For instance, yearly water use for hydraulic fracturing across the entire state of Pennsylvania is only 0.2% of the total annual statewide water withdrawals (Vidic et al. 2013), a tiny portion. Specific extraction locations, which may be clustered on particular streams, also have an effect on accurate measurement of water availability and use, a problem discussed further in Section 3.b.

The second scalar problem in research on water use in hydraulic fracturing is the mismatch between the measuring unit and the hydrologic unit. This not unique to hydraulic fracturing, and is common to many types of hydrological research (Rahm and Riha 2012). Water use is typically measured within administrative or political units—city, county, state, etc. From a governance perspective this makes sense: it is simple to assign responsibility for monitoring to an existing structure that operates within clearly-defined borders. However, this may result in an incomplete understanding of the complexity of local water systems: from the physical perspective, hydrological boundaries and conditions frequently overlap or are overlapped by several administrative jurisdictions; interact with different, transitory and transboundary groundwater and atmospheric systems; and vary temporally on seasonal and decadal scales. Administrative boundaries are rarely spatially consistent with the physical boundaries of hydrological systems. From the administrative perspective too this may not be ideal: available expertise may vary at different levels of government and between governments at the same level, resulting in uneven understanding and enforcement in neighboring jurisdictions. Again, these possible mismatches are not unique to research on hydraulic fracturing (Hussey and Pittock 2012; Bazilian et al. 2011), but they are germane: the EPA has recently identified that impacts of water use at smaller spatial and temporal scales can result in a substantially different understanding than existing, larger scale research has permitted, indicating the need for more local-scale analyses of hydraulic fracturing (EPA 2015).

This research will attempt to reconcile the impacts of the MAUP in two ways. First, this study disaggregates the research from the state level to the watershed level to determine the importance of variations between cataloging units. Second, this research uses standard watershed delineations that have been developed and used by public agencies for water management. Specifically, this research uses the eight-digit hydrologic unit watershed (HUC-8,[8] also known as a cataloging unit) as the spatial scale of analysis. Using this scale, for which decades of historical data are available, will allow clear comparisons between relatively small and numerous watersheds (55 in Pennsylvania alone; see Map 4.1) at a much higher resolution than existing studies. The

[8] The HUC is a method of naming individual watersheds in the United States. Each HUC is identified by a numerical code in which an increasing number of digits indicates a smaller watershed. For example, "02" is the Mid-Atlantic Watershed, "0204" is the Delaware River, "020401" is the Upper Delaware River, and "02040103" is the Lackawaxen River, a tributary in the upper portion of the Delaware River.

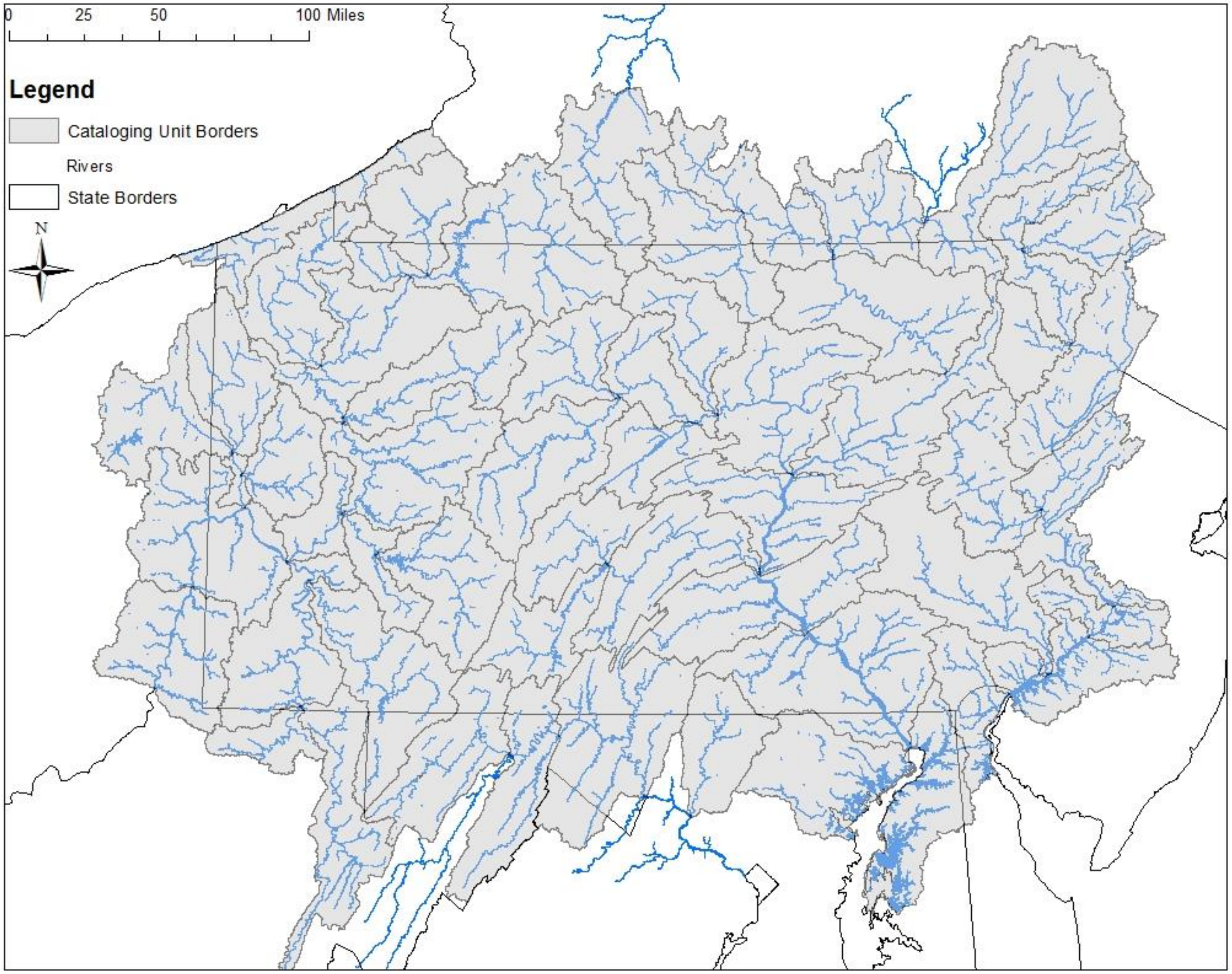
US Geological Survey (USGS), which created the HUC numbering system, labels watersheds to the HUC-12 level, but discharge volume is not consistently measured below the HUC-8 level.

Using the watershed level to measure water availability carries the risk of concealing impacts on water management from differing legal regimes emplaced by state-level administrative bodies. Most regulation of water resources and the oil and gas industry is at the state level, though it is loosely guided by the federal government, in part through the Safe Drinking Water Act (Hannah Jacobs Wiseman 2009). In order to control for these differences, this study will focus on those cataloging units that are partly or entirely in Pennsylvania. Cataloging units overlapping state borders are also included. The majority of ongoing HVHF in Pennsylvania takes place at the edges of the state in the southwest and northeast parts of the state, though HVHF operations are expected to expand further into Pennsylvania's borders in the future. Excluding border regions would thus exclude the major centers of expected HVHF (and presumably, water extraction). Water extraction in Pennsylvania is coordinated by the Pennsylvania Department of Environmental Protection (PADEP) and the applicable river basin commissions (Delaware River and Susquehanna River Basin Commissions; the Allegheny River basin does not have a formal compact like the DRB and SRB, so is managed by PADEP). These commissions are federally-established compacts that include stakeholders from all states traversed by the rivers.

This research will model water availability on a daily basis, aggregate it to seasonal flows, and further aggregate it to model the volume of water available in an average year. Modeling water on a daily basis follows the available data, which is measured in daily average cubic feet per second. Average daily discharge is also an important input to the process used to calculate water withdrawals during periods of low flow (SRBC 2012). Aggregating daily discharge to seasonal discharge smooths daily variations, allowing for more accurate comparisons—comparing a stream's daily average discharge volume for a given day in the spring (a generally high discharge period) to the average daily discharge volume in the late summer (a generally low discharge period) would result in enormous inherent variation because of high daily flow variability. Comparing average discharge volumes between two spring seasons makes it possible to identify differences between periods that should be similar from year to year. By combining the seasonally-adjusted flow rates of each season, I am able to model an average year while accounting for seasonal variations. Measuring water availability on a yearly scale without accounting for seasonal variation may be sufficient for broad overviews; however, these seasonal changes are critical to this research because asserting that a certain amount of water may be withdrawn each year conceals great variation in when it can be withdrawn.

It is necessary to define the seasons in question in order to measure differences in seasonal discharges. Maine's water managers, who live in an area with a similar climate to Pennsylvania's,[9] divide a year into six seasons (winter, spring, early summer, summer, fall, early winter), and measure the "seasonal aquatic base flow" (SABF), defined as the median flow in the later of two months in each of its six seasons: February, April, June, August, October, and December (Maine DEP 2007). This metric is applied to identify a

[9] Both states are mostly Koppen Dfa climates (humid continental), though parts of southern Pennsylvania are considered Dfb (similar, but with cooler summers than Dfa climates).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Cataloging Unit Borders
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N

Map 4.1 Cataloging units used in this research.

long-term historical seasonally-adjusted aquatic base flow (H-SABF) for each stream in each season using historical daily mean discharge data.

According to Richter et al. (2012) there are two bounds for “sustainable” streamwater extraction: 10% of discharge and 20% of discharge, based off of a river model that is updated daily given a number of conditions. Up to 10% of that modeled flow can be extracted and the “natural structure and function of the riverine ecosystem will be maintained with minimal changes.” A reduction of up to 20% results in unspecified but “measurable changes in the structure [of the stream] and minimal changes in ecosystem functions.” Reductions of greater than 20% will “likely result in moderate to major impacts in natural structure and ecosystem functions.” This method of determining safe extraction would be highly precise, but it faces three practical problems: it requires a massive investment in measurement infrastructure, personnel, and communications. In order to put such a system in operation, “natural” stream discharge would have to be determined over a period of years and the ability of existing infrastructure (dams and reserviors) to affect streams by increasing or decreasing flow must be modeled for each dam and reservoir. If models were to be run on a daily basis, information would also have to be disseminated on a daily basis to all water users. While not impossible given today’s technology, it would require widespread social acceptance, no doubt enforced by some legal mechanism. Finally, it is not at all clear if Richter’s modeling process would take account of the cascading effects of upstream extraction as discussed in the previous chapter. Failing to do that would make such model at best locally useful, and ill-prepared to be linked into watershed monitoring systems at increasingly larger and more complex scales.

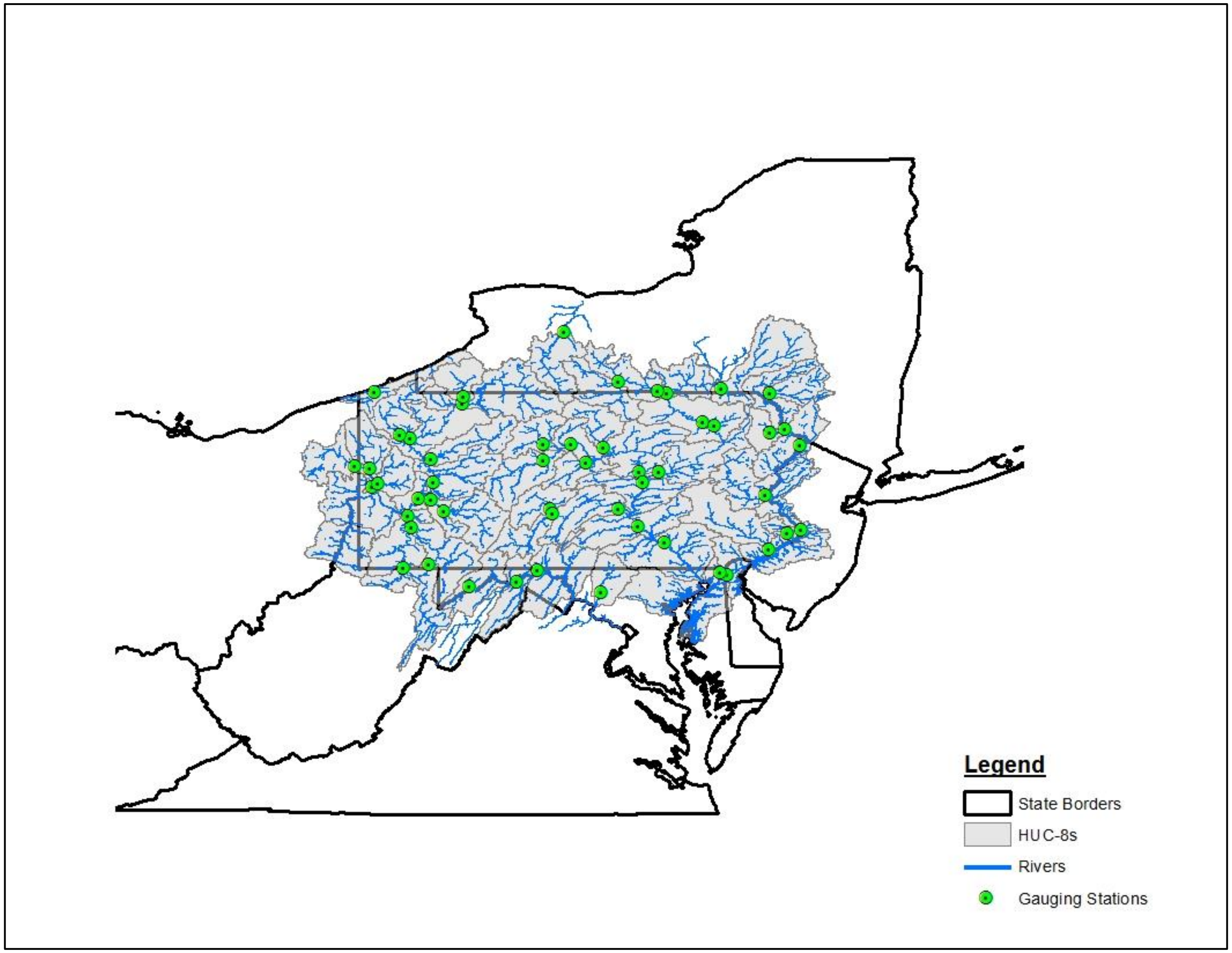
Thus, using elements from both the Maine DEP and Richter et al, a sustainable level of extraction is defined for this research as extraction that does not damage the riverine ecosystem while accounting for seasonal differences. The 10% and 20% levels of extraction will be used, but rather than basing them off of modeled daily flows, the reductions will be taken from the H-SABF. It should be noted, however, that individual streams may be more ecologically sensitive or may have other restrictions on water withdrawals, such as during popular recreational seasons or wildlife spawning times. Withdrawing even 10% of the stream flow in these locations could be prohibited at certain times.

Gauging Sites

Selection

The USGS has a vast network of surface water (and other) gauging stations, sometimes run in partnership with state and/or corporate entities. These gauging stations take in-stream measurements of data ranging from concentrations of various minerals, conductivity, and turbidity to temperature, pH, and most importantly for this research, discharge volume. Not all gauging stations collect all types of data, and not all stations run continuously. This research is focused on determining sustainable discharge volume, so when selecting gauging sites for this research, I first narrowed the universe of possible gauging stations to the set of stations that measure discharge volume. Reasoning that a sustainable flow at the confluence of a smaller river and a larger one would mean a sustainable flow throughout the cataloging unit,[10] I chose the gauging station that measured discharge as far downstream in the cataloging unit as possible (see Map 4.2). The simplest way to select the most appropriate gauge at which to take discharge data was to work backward from larger streams to smaller ones (see Figure 4.1) using the USGS's National Water Information System Mapper (NWIS Mapper),[11] an interactive online tool showing the locations of all surface water, groundwater, spring, and atmospheric gauging sites. For example, to find the best gauging site for the Schuylkill River, a HUC-8 in the Delaware River subregion and Lower Delaware basin, I followed the Delaware River from Delaware Bay to its confluence with the Schuylkill River and then traced the course of the Schuylkill River upstream to find the first gauging station (that is, the one closest to the Delaware River) that measured discharge. Using this gauging station has two advantages, both of which maximize the amount of flow recorded. First, selecting a gauging station as far downstream as possible allows for the highest possible number of tributaries to add their discharge volume to the main stem of the cataloging unit before measurement; second, using this gauging station ensures that maximum land area, and thus potential runoff, in the cataloging unit is accounted for in discharge volume. In cases where the most downstream gauging station did not measure discharge, a sub-optimal gauging station was selected.

[10] This reasoning will be discussed further in "Stream Discharge: Boundary and MAUPs."
[11] http://maps.waterdata.usgs.gov/mapper/index.html. This is also part of the method used by the Susquehanna River Basin Commission (SRBC) to estimate low flow (SRBC Low Flow, 2012).



Map 4.2 Locations of gauging stations used in this research.

Most of the cataloging units used in this research are completely within Pennsylvania, and so site selection and data collection for these proceeded as described above. A different approach is needed, however, for the cataloging units straddling the border of Pennsylvania and its neighboring states. No cataloging unit precisely matches the state border, even on the Delaware River, which forms the boundary between Pennsylvania and New Jersey. Three situations are possible on the state borders: streams and the cataloging units are mostly upstream from Pennsylvania, mostly downstream, or they may repeatedly cross the border. When a cataloging unit is generally upstream from Pennsylvania, such as the Youghiogheny River, the same method is used as described above. The flow as the stream crosses the Pennsylvania border is assumed to take into account all upstream activity, and so can be measured as close as possible to the next cataloging unit. This carries the risk that an existing, unsustainable level of extraction upstream will be included in my calculations of purportedly sustainable withdrawals. This would result in any additional extraction, no matter how small, having detrimental ecosystem impacts. This issue will be discussed further in Section 3.b., but in the absence of evidence to the contrary, I will assume that cumulative existing upstream extraction is sustainable. For cataloging units generally downstream from Pennsylvania, such as the Monocacy River in Maryland's part of the Potomac River basin, either the last possible Pennsylvania gauging station or the first possible out-of-state gauging station was used, with the former preferred. This allows me to identify the impact of extraction within Pennsylvania borders, minimizing or eliminating the impact of extraction in the downstream state. Cataloging units that are part of a series of cataloging units both upstream *and* downstream from Pennsylvania (for example, the Tioga River, which rises in Pennsylvania, flows north into New York, then joins the Chemung River and flows back south into Pennsylvania) were measured as close to their confluence with the next downstream HUC as possible even if this was not in Pennsylvania. Excluding such rivers from my analysis would not only leave gaps in data collection, but would adversely affect measurement for the downstream cataloging units despite possible extraction in Pennsylvania.

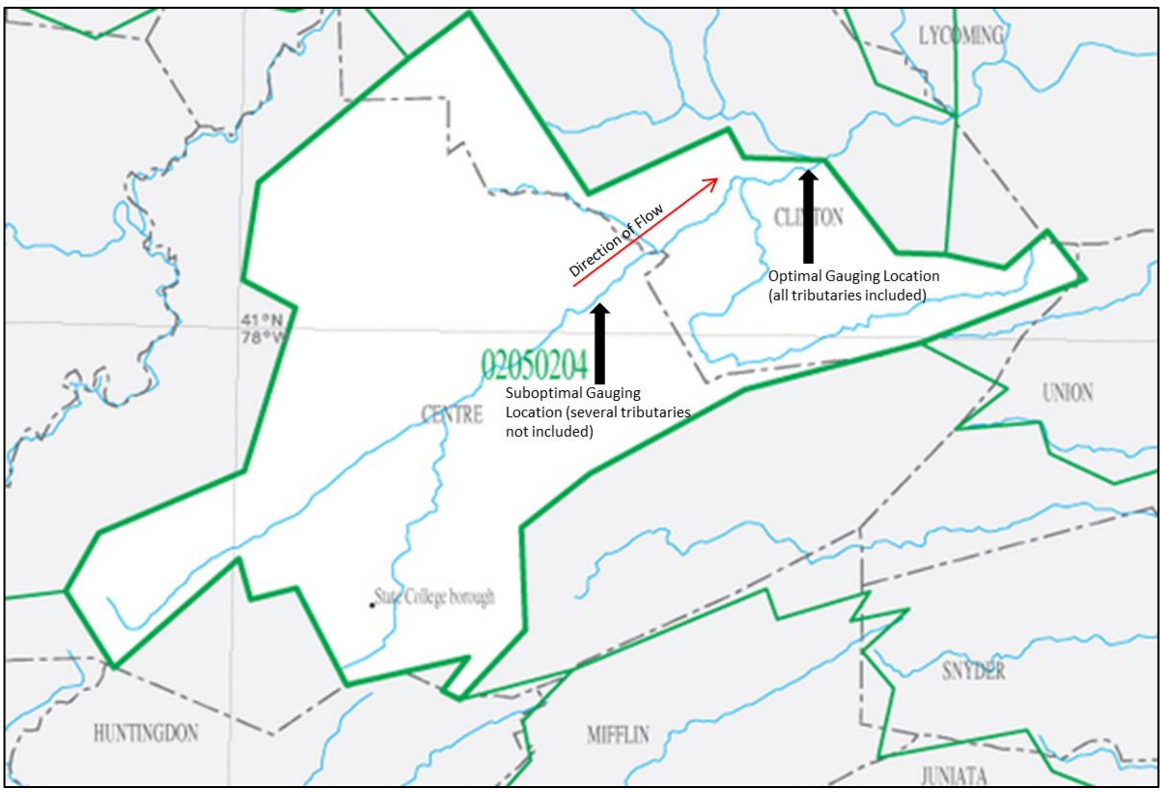


Figure 4.1 An optimal gauging location includes as many tributary streams as possible as well as the maximum land area of the cataloging unit. This allows for the gauging station to account for the maximum amount of streamflow and runoff respectively. A suboptimal gauging location does not include one or more tributary streams as well as large areas of the cataloging unit.

Except for West Virginia, there is little risk of states bordering Pennsylvania having an unexpected major impact on this analysis of Pennsylvanian rivers. Pennsylvania has six neighboring states (New York, New Jersey, Delaware, Maryland, West Virginia, and Ohio). Three of the states, New York, New Jersey, and Delaware, comprise a substantial portion of the Delaware River basin (New York also has a large part of the Susquehanna River basin within its borders). New York, the only state of these three with any shale resources, has recently prohibited HVHF,[12] but that is not the only major potential freshwater use in New York. New York City receives over half a billion gallons of water each day directly from the Delaware River basin and reservoir system via the Delaware Aqueduct. This value is approximately 50% of the total water used by the city each day.[13] With the population of New York City expected to grow to approximately 9.1 million by 2030 (NYC 2006) and therefore require more water, the New York Department of Environmental Conservation (NYDEC) embarked on a major renovation of the Delaware Aqueduct beginning in January 2013. Expected to continue through 2021, this project aims to fix leaks that waste up to 36 million gallons of water per day (about 7.5% of the aqueduct's daily volume), and will help to offset the water demand from population growth (New York City 2013). New Jersey, which makes up the opposite shore of the Delaware River from Pennsylvania, contributes a large portion

[12] Though water can be legally extracted in New York and transported to Pennsylvania for hydraulic fracturing, this likely only happens in border areas. The precise sites of this extraction, as with all water extraction locations, are not connected to any particular use, hydraulic fracturing or otherwise. See Section 4.d. for more information.

[13] Ensuring water quality for New York City was an explicit reason that New York banned HVHF in June 2015 (NYDEC 2015).

of the land area in the Delaware River basin. However, all but one[14] Delaware River cataloging unit in New Jersey is shared with Pennsylvania, meaning that any changes in water use in New Jersey will be reflected in my results. Finally, all of Delaware is downstream from Pennsylvania's portion of the Delaware River, and thus can have no impact on flow rates in Pennsylvania.

Maryland and Ohio both have shale energy resources, mainly in the western part of the Maryland and the eastern part of Ohio where these states overlay the Marcellus Shale. Because of an ongoing analysis of the possible impacts of HVHF, Maryland has an effective moratorium on hydraulic fracturing, though it this not official or permanent and a few wells exist (primarily for test and research purposes). If this moratorium were to change, water extraction for HVHF in Maryland would primarily affect the Youghiogheny River basin upstream from Pennsylvania, requiring reevaluation of the results of this study as they apply to the Youghiogheny River. The remaining rivers that are in both Maryland and Pennsylvania are generally downstream from Pennsylvania in the Potomac River and Upper Chesapeake Bay subregions, so additional extraction in those locations, none of which have shale resources, would not affect water availability in Pennsylvania. Ohio's portion of the Upper Ohio River basin lies generally downstream from Pennsylvania's, and of course cannot impact upstream cataloging units. Parts of eastern Ohio are upstream from the Allegheny River basin and overlie the Marcellus Shale, primarily in the Shanango River basin. There is some HVHF in this area; however, the long-term expected number of HVHF operations in this area is much

[14] The Cohansey-Maurice cataloging unit (HUC 02040206) discharges into Delaware Bay south of the Pennsylvania border.

smaller than in Pennsylvania because the shale layer is thinner and thus less productive in Ohio.

Of all neighboring states, West Virginia is the state that could impact this research the most. It has a large shale energy industry and its rivers are all generally upstream from Pennsylvania, meaning their flows are dependent on factors that cannot be controlled by Pennsylvania. The Monongahela River basin is the largest basin that is in both West Virginia and Pennsylvania (the Youghiogheny River has a few river miles in West Virginia, but mostly flows through Maryland before entering Pennsylvania). It is not governed by an interstate compact, so there is no formal transboundary river governance in that basin. As with all rivers generally upstream from Pennsylvania, I will measure all Monongahela River basin streams inside Pennsylvania wherever possible, ensuring that any changes in the discharge volume of that river before it arrives in Pennsylvania will be included in my calculations. This does, however, carry the risk, mentioned above, of including an already-unsustainable flow in my recommended sustainable levels of extraction.

Stream Discharge: Boundary and MAUPs

Underlying the selection of gauging stations is the assumption that, if the farthest downstream location has an environmentally-sustainable discharge volume, all upstream locations also will. This assumption presents an excellent opportunity for a brief discussion of the boundary problem, a common bugbear of spatial analysis, as it relates to stream discharge. It also provides an opportunity to expand on the earlier discussion of the MAUP. For clarity, I will refer to this formulation of the boundary problem as a "stream discharge" boundary problem. At its core, it is caused by the layered and nested nature of watersheds of all scales which, when viewed on a map, have clear, topographically- and hydrologically-defined boundaries.

This stream discharge boundary problem flows from the fact that withdrawing water from one cataloging unit indirectly withdraws it from all downstream cataloging units before it can contribute to their flow. This lowers the discharge volume of all downstream cataloging units from their normal "untouched" levels despite the fact that no water was withdrawn from inside the boundaries of those downstream cataloging units. The exact reduction (and thus its importance), however, depends on the proportion of the tributary's discharge volume relative to the volume of the main stem. When withdrawing water from a downstream cataloging unit, one must always account for upstream withdrawals in order to ensure sustainability. Failing to do so would result in a greater proportional reduction than originally intended. One way to solve this problem would be to determine the proportional contribution to the main stem from each tributary, then reduce the allowable extraction from the main stem by a value corresponding to the unaltered flow minus the sum of the total extraction in upstream cataloging units. Accurately doing so would require a large number of precisely-placed gauging stations, but it is possible to estimate this using the gauging stations here. Figure 4.2 and Table 4.1 show a simplified watershed and the possible impacts of upstream water extraction. This thesis will model the impacts of upstream extraction by determining the proportional contribution of each upstream cataloging unit to each downstream one. Next, I will model extraction in the direction of river flow (upstream cataloging units followed by downstream ones). This finds the reduction to flow in downstream cataloging units, and therefore the “new” allowable downstream extraction volume. This is based off of the fact that not all tributaries contribute equal volumes to downstream cataloging units—withdrawing 10% of flow from an upstream cataloging unit that only provides 10% of the downstream flow has less impact than withdrawing 10% of flow from a cataloging unit that provides 90% of the downstream flow. Coincidentally, many of the cataloging units in Pennsylvania expected to be highly exploited for shale fuels[15] are also in the upper reaches of the basins that contain them, and so this method of modeling provides a good approximation of actual conditions.

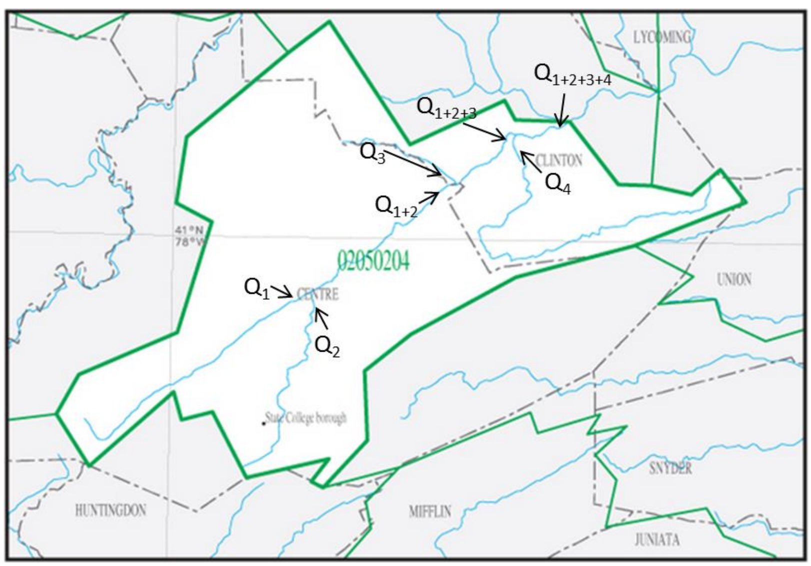


Figure 4.2 USGS view of Bald Eagle Creek, Pennsylvania, with hypothetical gauging locations added by the author. Combined with Tables 4.1 and 4.2, this shows the problem arising from the layered nature of cataloging units. While only 100 units of water were withdrawn from the main stem of the creek (at Q1), the discharge at Q1+2+3+4 has been reduced by 10% before any water was extracted at that gauging station because of the upstream water withdrawals.

[15] In general, the Upper Susquehanna River accounting unit (north-central Pennsylvania) and the Upper and Lower Monongahela River cataloging units are expected to be the most intensely-developed areas of Pennsylvania’s Marcellus Shale.

Table 4.1: Hypothetical numerical values for Figure 4.2. The top half of the table shows the original flow of the main stem (Q1) mixing with downstream tributaries and making up a smaller and smaller proportion of the flow of the main stem. The bottom half of the table shows that despite the withdrawal of 1% of the main stem's flow directly, the total extraction at the end of the cataloging unit is 10% of the expected volume because of extraction from the upstream watersheds.

Gauging Station	**Individual Q**	**Contribution to Main Stem Flow**			
		At Q1	**At Q1+2**	**At Q1+2+3**	**At Q1+2+3+4**
Q1	1000	100.0%	25.0%	12.5%	10.0%
Q2	3000	0.0%	75.0%	37.5%	30.0%
Q3	4000	0.0%	0.0%	50.0%	40.0%
Q4	2000	0.0%	0.0%	0.0%	20.0%

Gauging Station	**Q at Gauge**	**10% Q Extracted Upstream**	**Actual Contribution to Discharge at Q1+2+3+4**	**% Extracted of Q1+2+3+4**
Q1	1000	100	900	1.0%
Q2	3000	300	2700	3.0%
Q3	4000	400	3600	4.0%
Q4	2000	200	1800	2.0%
Total	10000	1000	9000	***10%***

At this point, we must again consider the MAUP because this research measures water availability at one site and uses that site to represent water availability throughout the entire cataloging unit. As discussed above, such aggregation from point data to areal data results in a loss of accuracy, in this case because water extraction probably does not occur exclusively at the gauging site. Instead, water extraction likely takes place in many locations across the watershed, some of which may be close to each other and therefore have outsized local impacts that may not be apparent at the level of the cataloging unit. As with the boundary problem, extraction from a first-order stream would impact the cataloging unit as a whole to an unknown extent. Again, sufficient gauging stations do not exist to measure all first-order streams and in any case, specific permit data would be required to determine precise water extraction locations (and therefore the likelihood and impacts of clustering). With HVHF likely continuing in Pennsylvania a decade or more into the future, many of those permits do not yet exist. Predicting their locations would introduce considerable uncertainty into the analysis, and as such an assumption has been made of an even rate of water extraction across the area of the cataloging unit upstream from the gauging station. However, to attempt to ensure sufficient downstream discharge exists, I will compare 10% and 20% outflows of the farthest downstream HUCs to the total upstream extraction.

Methods

Research Question #1

What is the sustainable rate of water extraction in each USGS cataloging unit in Pennsylvania, accounting for seasonal discharge variations?

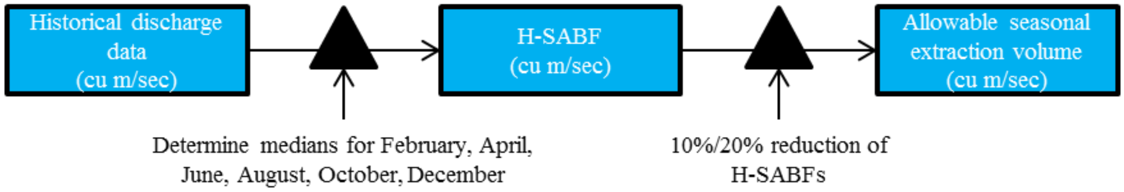


Figure 4.3 A simple visualization of the process of answering Research Question 1.

Streamflow data were collected using publically-available USGS data, as described in Section 3, with the exception of the Upper Ohio-Wheeling (HUC 05030106) and Upper Ohio (HUC 05030101) cataloging units, which did not have gauging stations that recorded discharge; these cataloging units were excluded from analysis. Daily data were taken as far back as was available. Every river had decades of data available, with one location (the Juniata River near Huntingdon, PA) having daily data available dating to 1891. Data from several rivers did not carry through to the present (records for the Shanango River stop in 1933, Bald Eagle Creek in 2000, the Clarion River in 2002, the Monongahela River in 2004, and the Lehigh River in 2005), and so results for these rivers should be reevaluated when more recent data are available. Though these rivers did not have data for recent years, all of them had several decades of data available for analysis, sufficient to account for decadal-scale climatic cycles. However, this lack of present-day data means that results for these rivers are uncertain and should be re-evaluated when more data is available to determine impacts from changes in extraction, climate variability, and other factors.

Data for each river were saved in a separate Microsoft Excel file to create easy-to-manipulate tabular outputs, and edited in minor ways to ensure functionality (ensuring numbers were not stored as text, for instance). The raw data taken from the USGS were provided as daily averages of cubic feet per second (ft^3/sec) which were converted to m^3/day. Most rivers had several days throughout their history on which the average flow volume was shown as "I" (Ice). This indicated that the gauge had been frozen on that day. According to the USGS, in these cases, gauges are temporarily disabled to prevent erroneous discharge values. Because the actual discharge value can only be known with extensive research,[16] I removed all such instances from the data set to avoid biasing median streamflow toward zero. Finally, I sorted the data by month and filtered each column.

Before modeling extraction from these gauges, I attempted to determine whether or not a long-term trends exists in each cataloging unit. To do this, I compared seasonal data for all cataloging units (35 of 55 total in this research, or 63%) with complete data for both 1970-1979 and 2004-2014. The 1970s were selected because they were the wettest decade since 1900 (Najjar 1999), while the period 2004-2014 is simply the most recent decade.[17] Compared to the 1970s, Pennsylvania cataloging units currently have higher discharge in the winter (29/35 = 83% of cataloging units have more discharge), early summer (19/35 = 54%), summer (23/35 = 66%), fall (29/35 = 83%), and early winter (31/35 = 88%); only in the spring do more cataloging units have lower discharge than in the 1970s (26/35 = 74% have less discharge). If this trend of generally drier

[16] The full text of the warning reads: "ICE EFFECTS ON STREAMFLOW--During cold weather, ice effects on stage and discharge determinations at some stream gages are likely. Stream gages experiencing ice conditions will have the discharge record temporarily disabled to prevent the display of erroneous discharge values. The discharge record will resume when it is determined that ice conditions are no longer present. Adjustment of data for ice effects can only be done after detailed analysis." (USGS.gov)

[17] 2014 was used because it was the most recent complete year of data. At the time of data collection, 2015 had not ended, so it was excluded.

spring seasons and generally wetter other seasons, this research will likely over-estimate the impact of water use for most of the year but underestimate it in the spring.

In a separate worksheet in each river's Excel workbook, I determined median discharge volumes for that river for each season, using every daily data point from all the Februarys, Aprils, Junes, Augusts, Octobers, and Decembers in each data set, thus creating the long-term historical seasonal aquatic base flows (H-SABF). Following Richter et al. (2012) the H-SABF values for stream discharge were reduced by 10% and 20% from their "natural" flow rates to determine the volume of water that could be extracted sustainably. That value, the average available cubic meters of water per day in each season, is the required input for the second research question.

Research Question #2

Which Pennsylvania cataloging units, if any, are at risk due to increased HVHF under high, medium, and low water use and well pad/well density scenarios?

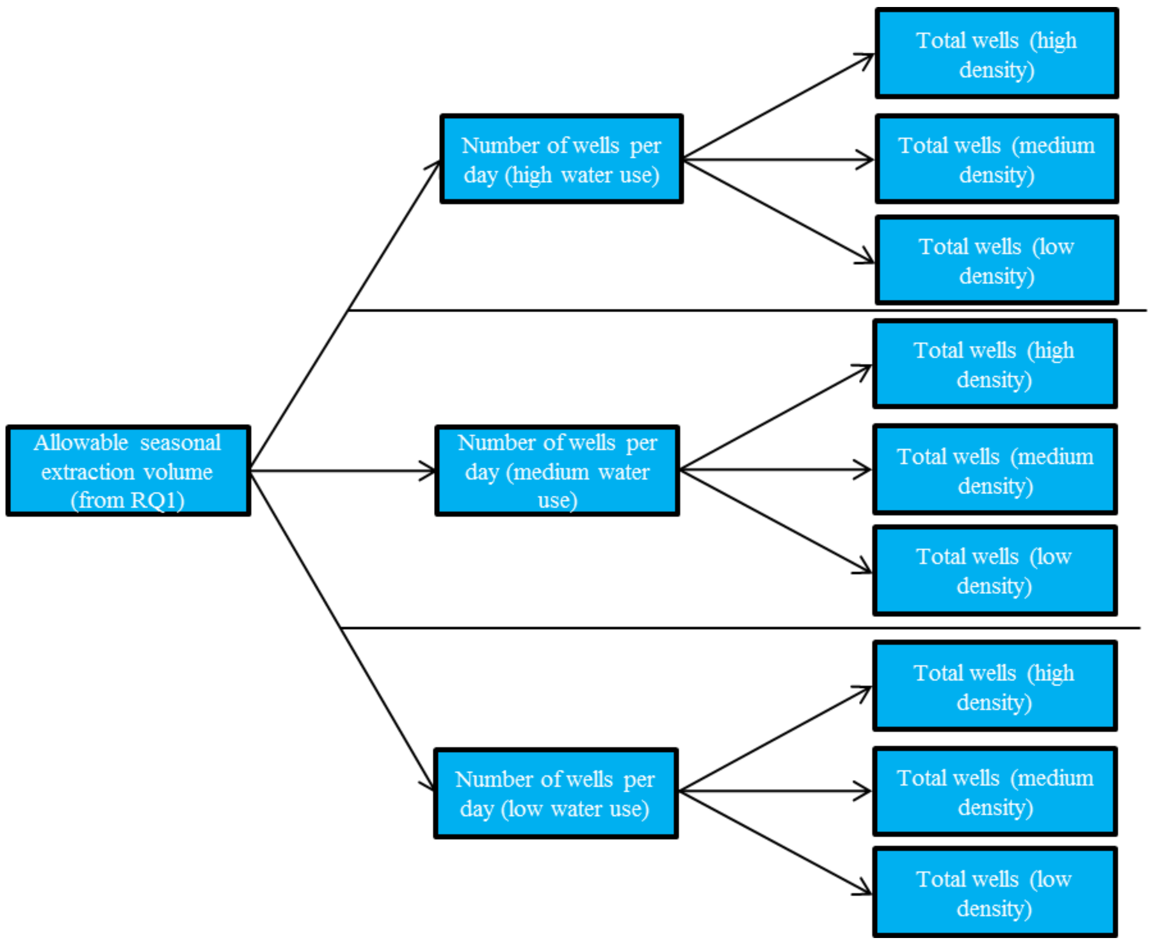


Figure 4.4 The process for answering Research Question 2. The input data for this step of the process is the output of RQ1.

Having determined the H-SABF and the amount of water available for extraction in every cataloging unit, the next step is to determine the number of shale wells that can be supported in each HUC while limiting water withdrawals at the 10% and 20% thresholds, and compare that to the number of wells expected to be created in that HUC. Key to this is accounting for changes to technologies and differing lateral lengths. These dynamics are captured through scenario analysis. Scenarios are determined using estimates of high, medium, and low water usage (indirectly reflecting changes described above), which were taken from literature (see Tables 2.2 and 2.3; also Table 4.2). The volumes of water used were chosen because they were specific to the Marcellus Shale. Low and high estimates of water use were simply the lowest and highest reported values, and the middle scenario is the median value of all the values researched (see Table 4.2). High development was defined as 15,000 new well pads across Pennsylvania each with four wells; medium development was 10,000 new pads with six wells each; and low development was 6,000 new well pads with 10 wells each.

Table 4.2. Scenarios. Values for well density are taken from The Nature Conservancy as described below. High, medium, and low water use estimates are taken from Rahm et al. (2013), Brantley et al. (2014), and Jiang, Hendrickson, and VanBriesen (2014) respectively.

		Water Use per well (m^3)		
		High	**Medium**	**Low**
Well Density (wells per pad)	**High**	4 / 30,000	4 / 15,142	4 / 3,500
	Medium	6 / 30,000	6 / 15,142	6 / 3,500
	Low	10 / 30,000	10 / 15,142	10 / 3,500

The final component of the model is flowback. Specifically, the model accounts for recycled flowback that might be used to offset some of the freshwater input. Peer-reviewed estimates for the amount of recycled water used in new wells range from 10% to 25% of the total injected volume with a median across the literature of 12% (n=8, only Marcellus Shale estimates used) (Rahm and Riha 2012; Mitchell et al 2013; Jiang, Hendrickson, and VanBriesen 2014; NETL 2013; see Table 2.3). For the high water use scenario, I assumed that 0% of the input volume for a well would consist of recycled water. In practice, most wells use at least some proportion of recycled water, although there is no requirement to use recycled water. Assuming that some wells use exclusively freshwater creates a theoretical maximum water use scenario. The intermediate scenario used a value of 12% for the proportion of recycled water used. The maximum proportion of recycled water used was 25%, based off of an estimate from NETL, and was used in the low water use scenario.

In order to adjust for seasonal water availability, I compared my 10% and 20% allowable extraction scenarios to the historical data to determine the expected number of days in each season that extraction would be prohibited. I assumed that extraction would be prohibited on all days when the otherwise allowable extraction volume exceeded the 10% or 20% threshold because of low discharge. To do this, I found the number of days that this was the case for each season and divided that number by the number of seasons of data available to find the average number of days (with a variance of two standard deviations) in each season that extraction would be prohibited.[18] Total yearly water availability was determined to be the sum of all daily H-SABFs (converted from m^3/sec to m^3/day) minus the foregone extraction on the number of days for which water extraction was prohibited.

The final step for the physical process is to determine, in the absence of regulation, if water requirements from projected rates of hydraulic fracturing compromise sustainable water extraction rates for any cataloging units in Pennsylvania. This portion

[18] For example, ten summers (Julys/Augusts) of data means 620 days of data (31 days in July + 31 days in August = 62 days per summer per year *10 years = 620 days. If 100 of those 620 days had a flow rate less than 10% of the H-SABF, extraction in an average year can be predicted to be prohibited 10 days per summer (100 total days/10 summers = 10 days per summer, or on average water may be withdrawn 52/62 days each summer).

of the analysis will map vulnerability to increased water extraction from hydraulic fracturing at the HUC level. Estimates abound for the amount of shale energy wells that will be drilled in Pennsylvania's portion of the Marcellus Shale. One of the most recent and comprehensive estimates comes from The Nature Conservancy, a nonprofit conservation organization that seeks to "protect ecologically important lands and waters for nature and people" (NatureConservancy.com). The organization was founded in 1951 and has branches in all 50 US states and 35 countries.

In 2011 a team of the Nature Conservancy's staff scientists estimated that 60,000 new shale energy wells would be established in Pennsylvania by 2030, and predicted their locations and the probabilities of those locations. In order to arrive at this estimate, the team assumed that 250 drilling rigs would be able to drill one well each per month, every month until 2030. They also assumed there would be stable prices, stable capital markets, and that recent trends of shale development would continue.[19] Their work resulted in high, medium, and low density scenarios based on varying numbers of wells per pad as well as probability estimates for where these wells would be. The Nature Conservancy mapped the expected locations of these new wells based off of a map of areas expected to be geologically productive; the resulting locations are therefore likely, but not guaranteed, and importantly, not based off of known future locations (for example, locations with approved permits). TNC overlaid these probable locations on administrative boundaries. By overlaying their map of expected locations with a map of HUC-8 watersheds, I am able to assess the impacts of projected activity at the

[19] Consistently low prices for oil and gas, such as have been seen since approximately the summer of 2015, have likely changed this calculus but it is too early to determine the elasticity of these prices and the long-term impacts on the shale oil and gas industry.

hydrological scale. A caveat to the analysis done by the Nature Conservancy is that increased density does not reliably increase the number of wells and well pads in a given cataloging unit. For example, the lower Monongahela River cataloging unit is predicted to have 11 well pads (44 wells) in the high density scenario, six pads (36 wells) in the medium density scenario, and 10 pads (60 wells) in the low density scenario. This does not change the method of analysis or the data, but it should be noted.

The cascading effect of upstream water extraction on downstream cataloging units is the final way streams may be vulnerable due to HVHF. In order to account for cascading impacts from upstream extraction, I determined the seasonal proportional contribution of each tributary to the next-largest river[20]. For all the cataloging units that were expected to host hydraulic fracturing, the frequency of allowable well installation (and therefore total number of wells allowed per season) is determined by the 10% and 20% flow extraction threshold values." For example, if 10,000 m^3 of water flowed through a cataloging unit each day and 30,000 m^3 are required to create a well, one well can be created every three days. A cataloging unit expected to host 10 wells would therefore take 30 days to allow sufficient flow to create all the wells, or less than one season. Subsequent seasons would see zero extraction, because the water requirements for drilling and extraction had already been met in the cataloging unit. This is an extreme scenario—it is unlikely that 10% or 20% of a particular stream's discharge volume can be extracted reliably for weeks at a time. However, such an extreme scenario will provide an upper limit for extraction in a cataloging unit and show the cascading effects downstream.

[20] See section on "Stream Discharge, Boundary and MAUPs," p. 59.

My results rest on one assumption: wells in a given watershed will be created using surface water and recycled water. Sixty-seven percent of water for HVHF drilling in the Marcellus Shale in Pennsylvania comes from surface water, with the remaining 15% and 18% from recycled water and groundwater, respectively (EPA 2015). This surface water will likely come from sources in the same watershed. This is the genesis of my second boundary problem.

Water Transport: Boundary Problem #2

The "water transport" boundary problem is caused by the assumption, which I am unable to validate,[21] that nearly all fresh water being used for hydraulic fracturing is withdrawn from the same basin in which the well pad is located. This assumption is reasonable for one deceptively simple reason: water is heavy, and therefore expensive to move long distances or over the topographic highs of watershed boundaries in the large quantities required for hydraulic fracturing. At least 400 tanker truck trips are required to deliver water from the source to a well, assuming common 5,000 gallon (19 m^3) trucks are used. Truck transport is the most inefficient and therefore most expensive method of transporting large quantities of heavy goods for long distances. While it is not always possible to operate water trucks with perfect efficiency and trans-boundary water transfers do occur, the cost of longer trips makes it logical that shale well drillers would seek to minimize the distance (and therefore cost) of individual trips. By allowing a

[21] For example, the SRBC issues permits for water withdrawals and "monitors" (though does not issue permits for) shale energy development within the basin. Because of this, water withdrawal locations and shale well locations are not tracked together, so it is impossible to say that water taken from a particular source was used at a particular well, merely that water was taken, and a well was created.

smaller number of drivers to make many short “turns” to deliver water, the required quantity of water will be on site more cheaply than if a large number of drivers made only a few long “turns.” Similarly, pumping groundwater requires more complicated and therefore expensive infrastructure than simply withdrawing water from a river or a municipal waterworks. In most cases therefore, the assumption that nearby surface water is the primary source of input water will be correct; however this may change near the boundaries of cataloging units.

At these boundaries, the likely origin of fresh water becomes less clear. There, it may be closer, and therefore cheaper, to make an inter-basin transfer out of one watershed to a well site in another. That formulation of the boundary problem could have serious ecosystem consequences, since inter-basin transfers of water effectively remove water from one watershed permanently—all the water extracted is consumed, none of it is simply used. Even if 100% of the injected volume returns from a well (likely impossible, as we saw in the previous chapter) and is safely returned to the environment, the probability that it is returned to the original watershed is low—what oilfield services company would truck its wastewater back to the original extraction site? These net transfers may be visible in the results—a cataloging unit with a low discharge volume and a high density of predicted wells bordering a high discharge and/or low predicted density cataloging unit may be an indicator of transboundary water transfers.

Research Question #3

What is the state of the regulatory environment related to water extraction?

Physical limitations on water extraction—how much water may be extracted—are governed by civil bodies, which embody the perceived value of water to their members and (where applicable) the people they represent. These institutions can take many forms, including federal or state governmental bodies, regulatory agencies delegated power from those state bodies, and civil society institutions such as conservancy and lobby groups. In Pennsylvania, federally-established interstate river basin commissions are also important actors.

I will attempt to map the governance of water extraction for hydraulic fracturing in Pennsylvania to assess safeguards that might prevent over-extraction. My focus will be on institutions with legal authority, specifically river basin commissions, environmental agencies at the federal and state levels, and local governments. These organizations have decision-making power and in many cases, inspection, enforcement, and punitive authority. They also overlap spatially and therefore may present instances of regulatory fragmentation or duplication.

Regulations surrounding water extraction are important, but equally important is determining whether or not these regulations were adopted in response to stresses from hydraulic fracturing operations. Regulatory changes over time and in different areas are particularly relevant to determine whether or not hydraulic fracturing was a stimulus resulting in changes to the governance structure. Hydraulic fracturing began on a large scale in Pennsylvania in approximately 2007, and has primarily taken place in northeast and southwest Pennsylvania. Differences in laws and regulations, either statewide or within those areas, before and after 2007 may reflect the growing importance of hydraulic fracturing to Pennsylvania's economy and environment, as well as reflecting local perceptions. For example, one might expect to see counties in coal-producing southwest Pennsylvania react differently than those in historically unindustrialized northeast Pennsylvania. Such spatially and temporally different responses to the new stimulus of hydraulic fracturing may also have created opportunities for exploitation, which I will attempt to identify.

An institutional analysis of this type comes with inherent uncertainty, but provides an important starting point. Such an analysis can shed light on gaps and overlaps in institutions, and thus point a way to improving the regulatory frameworks surrounding water use in hydraulic fracturing. For example, executive institutions such as departments of environmental protection have mandates, but they may not be sufficiently equipped, funded, or have a sufficient number of trained inspectors to fulfill their mandate. Because of this, the effectiveness of even the most well-intentioned regulations could vary. Variation creates gaps that could be exploited, intentionally or not, and reduce the effectiveness of the system as a whole.

Chapter 5 Results and Discussion

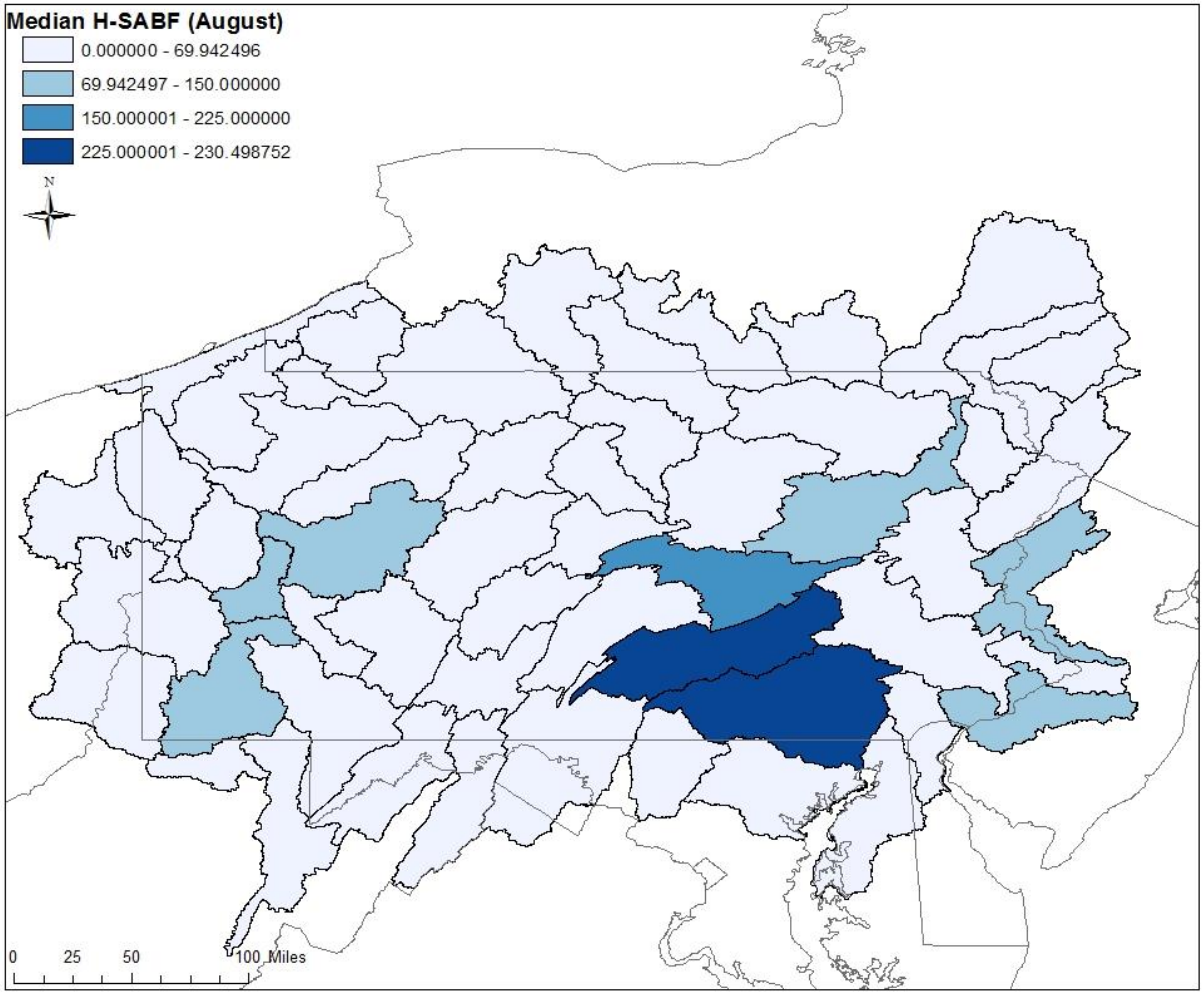
Results

Approximately 58% (32 of 55) of the cataloging units in Pennsylvania are expected to host hydraulic fracturing by 2030. As such, water use will continue to be a key issue for the future of the shale oil and gas industry, local populations, and environments and ecosystems across the state. Of the 32 cataloging units with projected shale development (Figures 5.1-5.3), those expected to host the largest number of wells are in the northeast and southwest parts of the state, already the areas with the most shale oil and gas development. The areas expected to host the least production (excluding the cataloging units predicted to have zero HVHF development) are in the Upper Delaware River in the eastern portion of Pennsylvania and face no risk to water resources.

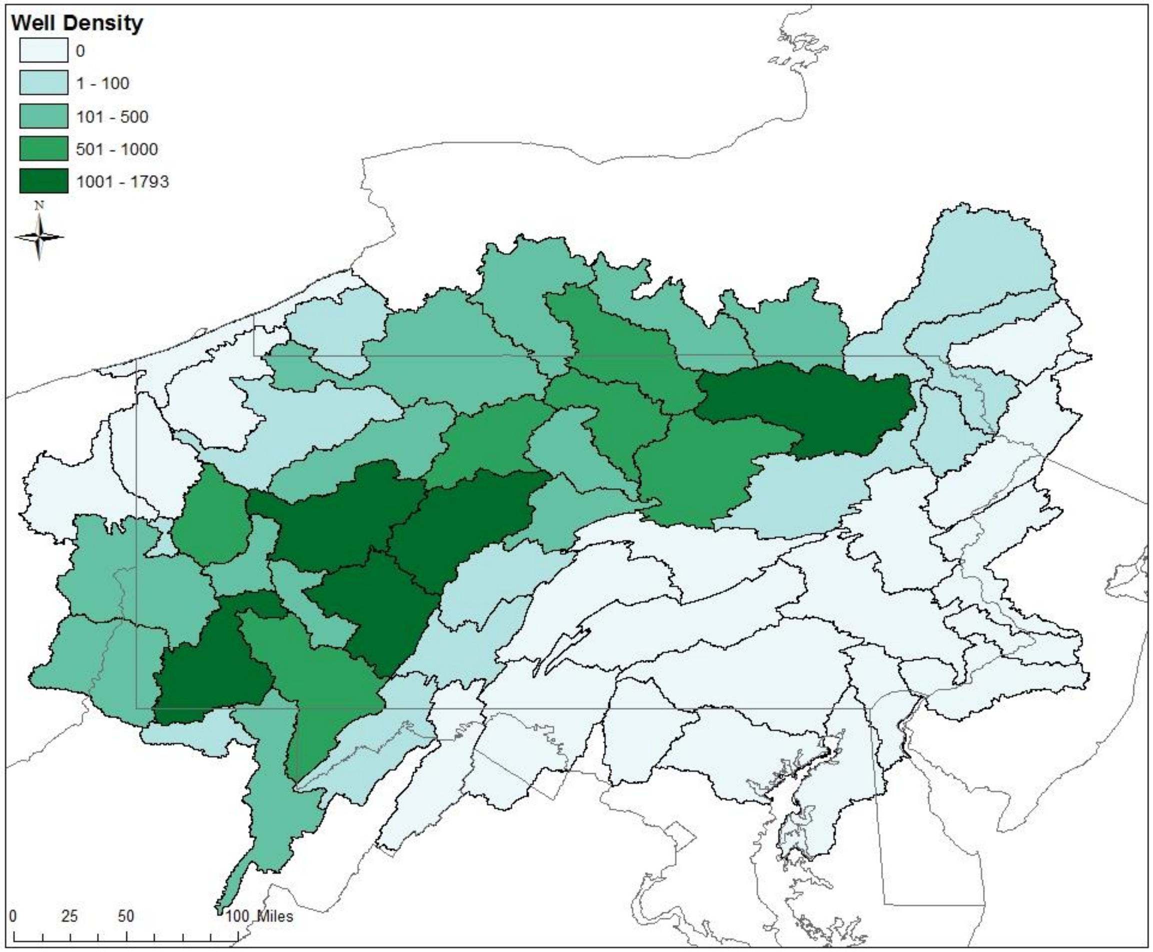
The Nature Conservancy predicted 60,000 wells would be created by 2030, allowing 19 years for development, four of which have already passed. Evaluating water availability for HVHF over that period of time shows that even if all water used for HVHF is taken from surface water sources in the same cataloging unit the well is created in, no cataloging units in Pennsylvania would be at risk. Water use efficiencies (including the use of reused/recycled water) and use of groundwater will further reduce any risk to Pennsylvania's surface water.

However, it is unlikely that HVHF operations will be spread as evenly as indicated in the Nature Conservancy research, which assumed similar conditions as existed in 2011. Nearly a year of a consistently low oil price has dropped the number of active horizontal drilling rigs to about 900 across the entire United States (Berman 2016). This slowdown in production may result in more intense production when oil prices rise again, raising the risk to water resources. Therefore, I analyzed the Nature Conservancy data based off of a "maximum extraction" scenario, defined as extracting water as quickly as possible, given a maximum limit of 10% of the H-SABF. Under this scenario, cataloging units were determined to be at direct risk if all HVHF operations in a cataloging unit require a constant 10% of the seasonally-adjusted discharge volume of the local cataloging unit for longer than one year. Cataloging units were only vulnerable under high water use/high density scenarios, indicating that efficiencies in water use can remove a large portion of the already-small risk of over-extraction. A total of six cataloging units, one each in the Allegheny River and Upper Ohio River accounting units and four in the Susquehanna River sub-region, required more than one year's allowed extraction volume to support all expected wells. Cascading effects were determined to exist when cataloging units without any predicted in-basin hydraulic fracturing operations were found to have less than 90% of the seasonally-adjusted discharge volume because of upstream extraction. Two cataloging units in the Lower Susquehanna River accounting unit were in this category. Four additional cataloging units, two in the Allegheny River accounting unit and two in the Upper Susquehanna River accounting unit may be at risk due to a combination of both factors (see Figure 5.4). These cataloging units are expected to support HVHF with stream discharge that has been reduced below its H-SABF due to upstream extraction.

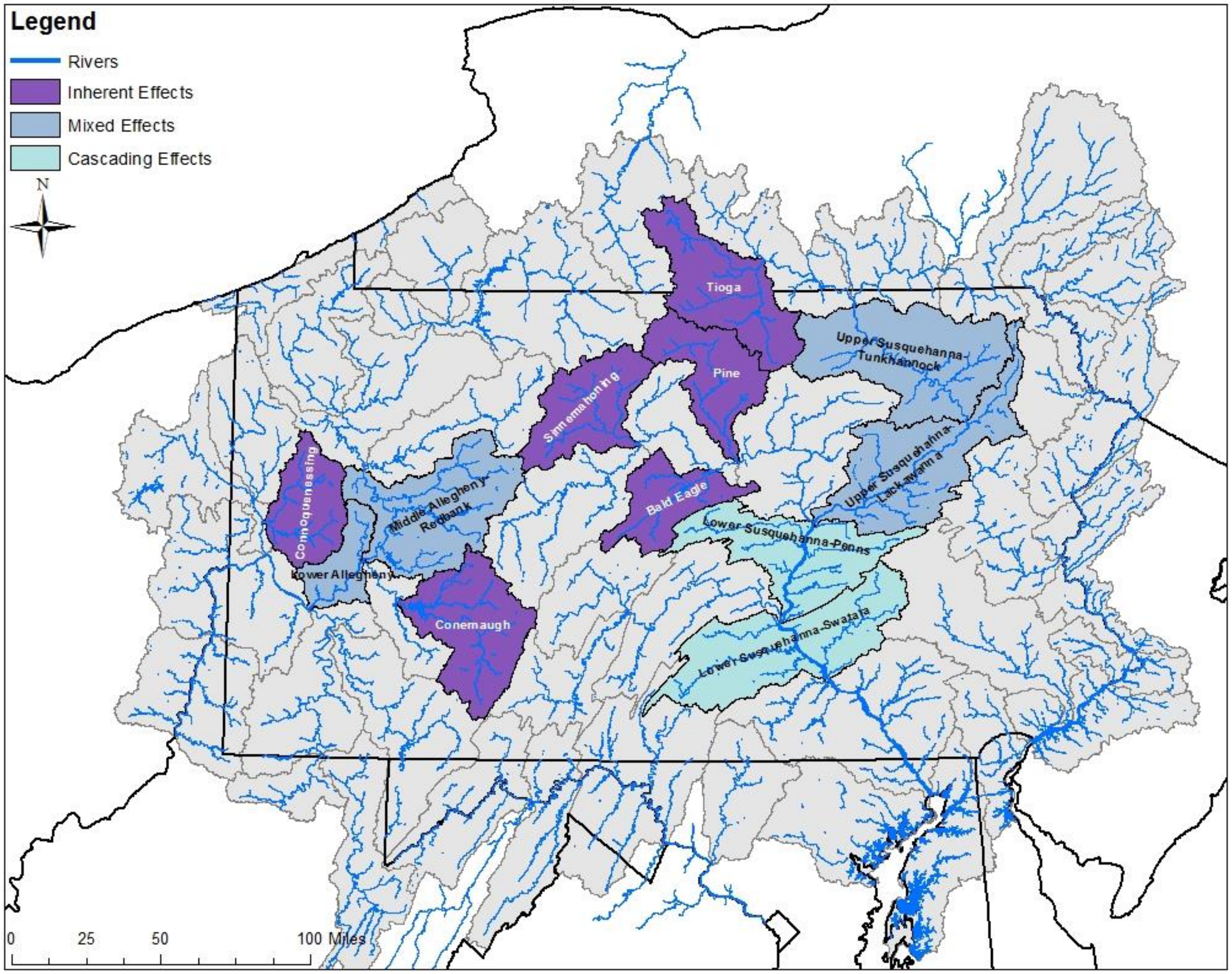
This research in effect measures imbalances between water availability and water demand for HVHF. The high, medium, and low scenarios present different amounts of wells in each cataloging unit, but to capture the highest possible water extraction, this research used the highest expected number of well pads/wells for each cataloging unit. With that in mind, the Connoquessing Creek cataloging unit in the Upper Ohio accounting unit is the most unbalanced in Pennsylvania. It is expected to host only the 10^{th}-largest number of hydraulically fractured wells of any cataloging unit in Pennsylvania (a estimated maximum of 2544), about 1/3 the number of the highest expected-development cataloging unit (the Upper Susquehanna-Tunkhannock cataloging unit, which covers a large area of northeast Pennsylvania). However, Connoquessing Creek's discharge volume is between the second- and fourth-smallest in Pennsylvania depending on the season (before accounting for upstream extractions in other cataloging units; Connoquessing Creek does not have any cataloging units upstream). At a consistent rate of extraction of 10% of discharge volume, it will take almost three and a half years to acquire sufficient water to hydraulically fracture all the predicted wells in that cataloging unit, so HVHF operations will need to be extended for a longer time than in other cataloging units, unless water is transferred from nearby cataloging units or groundwater is used. Barring such large inter-basin transfers or groundwater use, three and a half years likely underestimates the time required because that estimate assumed constant extraction every second of every day, likely far too optimistic. More likely, water extraction will take two or three times that long—simply assuming an eight-hour work day would triple the amount of time required for water extraction, with second- and


Median H-SABF (August)
0.000000 - 69.942496
69.942497 - 150.000000
150.000001 - 225.000000
225.000001 - 230.498752
N
0
25
50
100 Miles

Map 5.1 H-SABFs by cataloging unit for the month of August. Values are in m^3/s. August was chosen because it is the month used to measure summer discharge, and typically has the lowest flow, making rivers the most sensitive to major changes in discharge.



Map 5.2 Predicted numbers of wells in each cataloging unit based off of the high development scenario. Though the precise number of wells per cataloging unit changes in medium and low scenarios, at the cataloging unit level the relationships remain the same between the cataloging units.



Map 5.3 Watersheds deemed to be at risk due to inherent or cascading effects, or a combination of the two.

third-order effects on the rest of the process as well as downstream. The Conemaugh River, a tributary of the Allegheny River to the east of Pittsburgh, is expected to host the third-most wells of all cataloging units in Pennsylvania (4850), but its discharge is in the top half of Pennsylvania's streams (between 15th and 20th-smallest of 32 streams) and so is less imbalanced than Connoquessing Creek. Again assuming 10% extraction, sufficient water will flow through the Conemaugh River to allow for all wells to be hydraulically fractured in about 366 days.

Table 5.1 Maximum number of wells in each cataloging unit compared to statewide rank in seasonally adjusted flow.

			Statewide Seasonal Flow Rank (out of 55)					
Cataloging Unit	**Accounting Unit**	**Number of Wells**	**Winter**	**Spring**	**Early Summer**	**Summer**	**Fall**	**Early Winter**
Bald Eagle	W. Br. SusQ	1,500	43	39	38	35	37	41
Conemaugh	Allegheny	4,850	26	27	28	29	27	30
Connoqueness ing	Upper Ohio-Beaver	2,544	42	43	43	44	43	43
Pine	W. Br. SusQ	3,180	36	34	35	38	36	33
Sinnemahonin g	W. Br. SusQ	3,264	38	36	39	40	40	35
Tioga	Upper SusQ	3,188	40	35	36	39	39	36

Four cataloging units in the Susquehanna River accounting unit require more than a year's worth of allowable extraction volume to provide sufficient water for all expected wells. They are Bald Eagle Creek, Pine Creek, and Sinnemahoning Creek in the West Branch Susquehanna River accounting unit and the Tioga River in the Upper Susquehanna River accounting unit. Bald Eagle Creek will require slightly more than a year's worth of allowable extraction to supply all the water required for the wells expected in that cataloging unit. It ranges from the 5th to the 12th smallest cataloging unit in Pennsylvania, and its 1500 predicted wells would be the16th most wells. Sinnemahoning Creek and Pine Creek both require significantly longer to supply the required volume of water: almost 13 months for Sinnemahoning Creek and almost 14.5 months for Pine Creek. Sinnemahoning Creek is expected to be the 11th most-developed cataloging unit, hosting up to 2184 wells, and its flow varies from 6th to 11th-smallest in the state. Pine Creek has a slightly-higher discharge volume, varying from 7th to 13th-smallest cataloging unit in Pennsylvania, but it is expected to have the 7th-highest amount of hydraulic fracturing, with up to 3180 wells created in its cataloging unit. The Tioga River, already in one of the most intensely-developed areas of Pennsylvania's northeast, is expected to host as many as 3188 shale energy wells, the fifth-most in the state, and it varies from the 6th to 11th smallest cataloging by discharge volume.

Though these cataloging units should be carefully monitored, the linked, layered nature of watersheds means that water extraction upstream from a particular cataloging unit may have major impacts downstream, even in areas with no predicted shale wells. Pennsylvania has two cataloging units exclusively affected by upstream extraction: the Lower Susquehanna-Penns cataloging unit and the Lower Susquehanna-Swatara cataloging unit. The Lower Susquehanna-Swatara is the second to the last Susquehanna River cataloging unit in Pennsylvania, so it absorbs the impact of all upstream extraction in the Susquehanna River sub-basin. Depending on the intensity of upstream extraction, there could be a full year during which the Lower Susquehanna-Swatara cataloging unit experiences discharge volumes of as low as 81.4% of expected seasonal discharge due entirely to upstream extraction in the "high/high" water use/well density scenario. After one year of intense upstream extraction though, nearly all the discharge will be restored to its expected levels because almost all upstream extraction will have been completed. The Lower Susquehanna-Swatara cataloging unit has two direct tributaries: the Lower Juniata River and the Lower Susquehanna-Penns cataloging unit, so it is affected by extraction in both. The Lower Juniata River and its tributaries are expected to host only 456 wells between them and are all small to mid-sized streams that face minimal risk due to HVHF. The cataloging units upstream from the Lower Susquehanna-Penns cataloging unit are expected to host 26,544 wells between them, almost half of the total expected wells in Pennsylvania; unsurprisingly, such a large number of wells will have a large downstream impact. For the highest-discharge seasons (winter and spring), the Lower Susquehanna-Penns could experience substantial reduction in its flow rate due to the cumulative upstream extraction, possibly reducing flow to as little as 79.9% of the expected winter discharge volume if water is extracted as quickly as possible. This is the lowest figure this study found, and indicates a need to carefully monitor that watershed, even though it is not expected to host any shale wells itself.

Four more cataloging units should be monitored because of both in-basin extraction and the indirect effects of upstream extraction: the Upper Susquehanna-Tunkhannock, the Upper Susquehanna- Lackawanna, the Middle Allegheny-Redbank and the Lower Allegheny. These four cataloging units are expected to host shale well development, but they are primarily impacted by predicted upstream development. In the Susquehanna River sub-basin, the Tunkhannock cataloging unit covers a vast swath of northeast Pennsylvania and is expected to host a remarkable 7240 shale wells, more than

10% of the expected statewide total. However, it is also a very-large discharge cataloging unit, ranging from the 5th to 9th largest in the state. Basin-internal water use presents little direct risk, but this cataloging unit is downstream from some of the most intense predicted development in the Tioga River and Chemung River basins, so its flow is substantially reduced because of extraction there as well. Combined, this extraction has follow-on effects in the Upper Susquehanna-Lackawanna cataloging unit, which, despite the fact that it is expected to have only 200 wells, could experience discharge volumes as low as 82% of the expected seasonal volume because of upstream extraction. Interestingly, this means that four consecutive cataloging units (Upper Susquehanna-Tunkhannock, Upper Susquehanna-Lackawanna, Lower Susquehanna-Penns, and Lower Susquehanna-Swatara) could see substantially reduced discharge volumes due primarily to upstream extraction rather than basin-internal development.

This holds true for the Middle Allegheny-Redbank and Lower Allegheny cataloging units as well. Both cataloging units have large discharge volumes and can support the number of wells predicted for their basins (4662 and 1990 respectively), but they are both impacted by upstream extraction. The Middle Allegheny-Redbank and its tributaries are expected to host a total of 8188 wells, half of which are in the Middle Allegheny itself, resulting in discharge as low as 88.1% of the expected volume, albeit for just one season. The Lower Allegheny, which is immediately downstream from the Middle Alleghany-Redbank and its tributaries (not including the tributaries it shares with the Middle Allegheny-Redbank) are expected to host another 8772, so the Lower Allegheny's discharge could be reduced to as low as 81.3%, though again for only one season. The cascading effects end there, as the Lower Allegheny River meets the Lower

Monongahela River to form the Upper Ohio River, a cataloging unit with no expected shale wells in Pennsylvania, though a small number may be created in Ohio.

The results of this research show that water extraction for hydraulic fracturing can be conducted sustainably in Pennsylvania provided it is paced over multiple years. Pennsylvania is projected to host hydraulic fracturing operations in 32 of the 55 cataloging units that are partially or wholly in its borders. Assuming a consistent 10% extraction of discharge volume, as many as 12 cataloging units may be vulnerable to over-extraction of water.

Pennsylvania is a state with several high-discharge rivers, so in most cataloging units, withdrawals that may seem large in absolute terms are actually small in proportional terms. Upstream extraction certainly has an impact on downstream discharge volumes, and it is cumulative; the farther downstream one goes, the less one cataloging unit contributes to the overall flow, but the greater the total upstream extraction volume is. This is borne out in the results, which show that all cataloging units at risk from upstream extraction are in the main stem of rivers near where they end or where they exit Pennsylvania.

These results represent a maximum use scenario under which companies would seek to create wells as quickly and efficiently as possible, so spacing water extraction out over longer than a year would help alleviate any risks to water resources, even in areas with a large number of expected shale wells. Extending hydraulic fracturing operations over more year would make the development of shale wells more sustainable in water terms, and possibly in economic terms as well: longer-term development may help to alleviate boom-and-bust employment in the Marcellus Shale because a smaller number of workers, employed on a consistent basis would tend to bring more stability than a huge number of workers for several short periods. Even stretched over a period of years though, there may still be the need for inter-basin transfers, particularly in the case of Connoquessing Creek. The estimates of wells taken from the Nature Conservancy are based off of 250 rigs creating one well each month. That means a minimum of about 10 months of drilling in the Connoquessing Creek cataloging unit, if all rigs in the state operate in that cataloging unit and they all operate at maximum efficiency with sufficient input materials on hand. More than three years of continuous water extraction plus nearly a year of continuous drilling is ambitious: it would require the focus of an entire industry on one, without rest, with all equipment working properly, all resources immediately available, and more extremely optimistic assumptions. However, with little to no data on such factors as inter-corporation collaboration, working hours, required maintenance, resource timelines, I am unable to determine precisely how optimistic these assumptions are. Inter-basin transfers will likely be necessary if all these predicted wells are to be created before 2030.

Pennsylvania's rivers will likely not be forced to support such a high, consistent rate of extraction, due a number of economic, technical, and social factors. Regulatory safeguards at the federal, regional, and state levels are among the most important of these factors. Though these regulations are sparse (especially at the state level), they provide an opportunity to study the multilevel governance of hydraulic fracturing. Those state laws that do specifically mention water extraction tend to be very general, allowing the majority of water governance to take place at a technocratic, regional level in the Susquehanna River and Delaware River basin commissions. By simultaneously viewing the resource needs of hydraulic fracturing and the societally-established limitations on their use, we gain a fuller picture of the hydraulic fracturing system's interaction with hydrological systems.

States are the primary regulators of oil and gas operations, including the water resources involved (Wiseman 2014). Chapter 78 of the Pennsylvania Code is titled "Oil and Gas Wells." Created in 1987, it has been updated several times since hydraulic fracturing began, with hydraulic fracturing specifically in mind: in 2009, 2011, 2013, and 2014 (Section 78.1, 2014). Subchapter C, "Environmental Protection Performance Standards" is primarily concerned with "water resources," though every section focuses on water pollution instead of availability. Groundwater pollution is of particular concern, and there is no specific mention of surface water except when discussing pollution. The closest this law comes is in Section 78.51:

> "(a) A well operator who affects a public or private water supply by pollution or diminution shall restore or replace the affected supply with an alternate source of water adequate in quantity and quality for the purposes served by the supply as determined by the Department.
>
> (b) A landowner, water purveyor or affected person suffering pollution or diminution of a water supply as a result of drilling, altering or operating an oil or gas well may so notify the Department and request that an investigation be conducted."

While these sections provide a method of redress and protection for the landowner at the expense of the well operator, they lack awareness of the transboundary nature of surface (and ground) water resources. Diminution of a water supply may happen anywhere in a stream network, even far away from the perspective of a "landowner, water purveyor, or affected person." It may not be noticeable to those without special expertise and/or equipment (the difference between 1000 m^3/sec and 790 m^3/sec probably is not visible to the naked eye, but would be ecologically harmful), nor may it be the fault of a nearby well operator: as shown in the previous section, several cataloging units with no wells at all may be faced with diminution of their flow from many small withdrawal locations upstream. Further, diminishing water availability may contribute to more pollution: if a particular pollutant must be diluted to an environmentally-safe level and the stream discharge is lower than expected, the same amount of a pollutant would have an increased effect. Determining who is "at fault" in such a scenario (and therefore who is responsible for mitigating the difficulty of affected water users) may be virtually impossible, especially if all upstream operators were properly permitted. This indicates that the outsourcing of basin governance to regional institutions (such as RBCs) that have a broader view on the basin's water resources is an effective strategy.

Chapter 110 of the PA Code is titled "Water Resources Planning" but provides similarly little guidance on water extraction. Its major foci are clear from the subchapter titles: General Provisions (mostly definitions of terms used in the rest of the law), Registration, Reporting, Recordkeeping, Monitoring, and Water Conservation. All of these except the last define administrative standards: how to apply for a permit, how to report water extraction, how to record your extraction, and how to measure your extraction; nowhere is it defined how much water extraction is allowable, when, where, under which conditions, etc. Subchapter E, "Monitoring," provides the only volume-based metric in Chapter 110. It defines those who must receive permits and report water use as those

> "whose total withdrawal from a point of withdrawal, or from multiple points of withdrawal operated as a system either concurrently or sequentially, within a watershed equals or exceeds an average rate of 50,000 gallons per day in any 30-day period and each person who obtains water through interconnection with another person in an amount that exceeds an average rate of 100,000 gallons per day in any 30-day period shall measure or calculate"

The last subchapter, Water Conservation, does not actually discuss water conservation in a meaningful way. Rather, it identifies reporting requirements for people who have begun or want to begin a water conservation project. Chapter 110, despite its name, has very little to say about water resource planning but a great deal to say about water extraction reporting. One can assume that the data provided by this chapter are collected and monitored by some unknown institution, that permits are approved after understanding the other stresses on water in the applicable watershed, but that is not at all clear.

Table 5.2: List of protected water uses in PA that are applicable to all surface water bodies. Source: PA Code Chapter 93, Table 2

Symbol	Use
WWF	Warm Water Fishes
PWS	Potable Water Supply
IWS	Industrial Water Supply
LWS	Livestock Water Supply
IRS	Irrigation Water Supply
B	Boating
F	Fishing
WC	Water Contact Sports
E	Esthetics

All surface water sources in Pennsylvania are protected in accordance with Chapters 93 (Water Quality Standards) and 96 (Water Quality Standards Implementation) of the PA Code, and are required to provide sufficient quantities of water of sufficient quality for a number of uses (see Table 5.1). There is no explicit hierarchy of these uses (PA Code Chapter 93; PA Code Chapter 96). None of the highest development areas listed above have any state-mandated special limits on water extraction aside from those listed in Table 5.2. All rivers in this research are protected for the benefit of either cold water fish or warm water fish (depending on the river), but only one area has a designation not common to all rivers: Connoquessing Creek, specifically its headwaters, which are considered an area of "high-quality" waters according to Section 93.9w. This determination affects pollutant discharge rather than water extraction, and while there are a number of water quality standards the area must meet consistently, there are no special restrictions on water withdrawals because of a "high-quality" designation. Because water quality and quantity are linked, water withdrawal from this area could theoretically be limited to preserve quality, but there is no method identified for how this situation would be addressed. The absence of an Allegheny River Basin Commission means that there is no mandated basin-scale governance other than state law that would provide such a method. Even avoiding this area though, water withdrawal permits may be issued for areas downstream from the headwaters that the "high-quality" designation does not apply to.

The Delaware and Susquehanna River Basin Commissions are congressionally-established interstate compacts founded because of the recognition that "water and related resources are regional and important at various scales" and that "comprehensive, multi-purpose planning will enable the greatest benefit" from the rivers' resources

(DRBC Compact 1964).[22] Each commission has a representative from each state, appointed by that state's governor, and one from the US Army Corps of Engineers that represents the federal government. The commissions represent an institutional realization that rivers are transboundary subjects that must be governed cooperatively. Because of their interstate mandate, the RBCs have promulgated regulatory systems for water extraction with the entire watershed in mind, and hydraulic fracturing operators are generally treated the same as all other water suppliers.

The DRBC rules for measuring and regulating water use in HVHF have not been codified as is the case with the SRBC's rules (DRBC Revised Draft Regulations, 2011; 18 CFR, Sections 801, 806, 807, 808, 2012). However, there is a high degree of similarity between the DRBC's draft HVHF rules and the approved SRBC rules, even including similar wording in some cases, indicating that the SRBC rules were a template for the DRBC's. In this section, I will discuss the proposed DRBC rules and the approved SRBC rules as one, except when they differ substantially. The two RBCs regulate water consumption for all projects, including oil and natural gas development, in three important ways: approval processes, models of low-flow rates, and rules regarding intra- and inter-basin transfers.

As has been the case in most water regulations reviewed in this research, the focus for both commissions is on pollution and permit processes. Permits (or "docketed approvals") for water withdrawals are very specific. For the SRBC, a docketed approval includes a location (based off of the location provided on the permit request), maximum

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[22] Similar text from the SRBC Compact reads "In general, the purposes of this compact are…to provide for cooperative and coordinated planning and action by the signatories with respect to water resources." (SRBC Compact 1972)

volumes (in both gallons per day and gallons per minute) that may be withdrawn, reporting requirements, and limitations on water withdrawals based off of USGS gauges specific to the extraction site as well as modeled water levels.[23] These limitations are determined using a combination of flow exceedance values and Aquatic Resource Class, a stream sensitivity measure based off of the size of the watershed (in general, the smaller a stream is the more sensitive it is). Restrictions become stricter as watershed size decreases—for example, withdrawals from the main stem of the Susquehanna River may be limited when flow is at the Q95 level (only when the current discharge is exceeded 95% of the time) whereas for the smallest category of first order stream, water extraction may be limited at the Q5 level (only when the current discharge is exceeded 5% of the time) (SRBC 2012). This differs from most of the rest of the northeastern United States which uses a "Q 7, 10" model under which extraction is limited if the flow is expected to be below the 10-year expected low flow for seven consecutive days (Reilly and Kroll 2003). The SRBC's method, which combines gauge levels and models, is more easily tailored and modified than the "Q7, 10" method. While effective for modeling, the "Q7, 10" method may not be sufficiently accurate or reactive to keep pace with sudden changes in stream discharges. This is in part because it is not a "rolling" metric; rather it is only calculated irregularly, but typically every 10-15 years. The SRBC's addition of a gauge level requirement helps to mitigate the risks of pure modeling using empirical measurements in specific streams.

[23] All of these are requested on the initial permit application but may be changed unilaterally by the commission in order to integrate new permits with existing ones. However, changes are intentionally as small as possible compared to the permit request while still maintaining SRBC water resource goals (SRBC, personal communication).

Permits for water withdrawals in the SRB and DRB for unconventional oil and gas extraction follow generally the same process as those for other uses, but they differ in one important way: their renewal timelines. Both the SRBC and DRBC require all projects, not just HVHF ones, to commence within three years of approval or start the permitting process anew. This allows the commissions to ensure expected water withdrawals can be continuously modeled, monitored, and updated. However, while withdrawals for purposes other than unconventional oil and gas extraction require renewal every 15 years in both jurisdictions, permits for water extraction for unconventional oil and gas must be renewed every five years (DRBC 2011; 18 CFR 2012). Given the rapid rise of HVHF, this relatively speedy reevaluation is smart: with thousands of wells projected to be created each year, intensive water extraction spread over a period of years could have unknown impacts. Though most cataloging units In Pennsylvania are not at risk of over-extraction according to the H-SABF method, all the measurements taken for this research were taken from relatively large streams. Smaller streams could be altered substantially with even small amounts of extraction, and those effects, as we have seen, could cascade into much larger ones downstream. While a five-yearly renewal requirement is good, an annual reevaluation of permits might be better, ensuring that every stream is monitored closely, and having the salutary effect of keeping the RBCs informed of the yearly number of wells created in their area.

The final important water governance issue in the Susquehanna and Delaware River basins is intra- and inter-basin transfers. Transfers into or out of the basin—from the Susquehanna to the Delaware or vice-versa for instance—must be approved in a process similar to the permitting process. However, there are no practical limitations on intra-basin transfers. That is, transfers between, for example, Pine Creek and the Tioga River are not recorded, even though the former is in the West Branch Susquehanna accounting unit and the other is in the Upper Susquehanna accounting unit. When water extraction is approved at a particular location, there is no record of where that water is taken. Logically, companies would seek to minimize the transport for reasons described in Chapter 4, but that remains an assumption without any permit data to reinforce it. For all the benefits brought by regional-scale river basin governance—a holistic look at an entire basin, collaborative decision-making, a focus on a particular resource—they do not consider transfers inside the particular basin to be notable (SRBC personal communication 2015; DRBC 2011). This regime could present a terrific opportunity to study the water transport boundary problem discussed above, in the real world, at the regional scale. One wonders how much water is transferred between basins, and from where to where. Unfortunately, at the cataloging unit scale used in this study, the lack of data makes it impossible to answer how much water is transferred between cataloging units inside a river basin, leaving that question unanswered for now.

Summary of Results

Pennsylvania's water-energy nexus has changed in one important way because of the introduction of HVHF: there has been a shift from water use to water consumption that has not been reflected in legal frameworks. Thermoelectric power needs massive amounts of water for cooling, accounting for 66.2% of Pennsylvania's water withdrawals in 2010 (USGS 2014). Hydraulic fracturing, by contrast, uses a relatively small amount of water compared to thermoelectric power: each day, all mining operations (not just HVHF) use about 1.1% of the volume of water used for thermoelectric cooling (though this varies by location). However, more than half the amount for thermoelectric power was for once-through cooling, meaning that about 1/3 of the water withdrawn from Pennsylvania's rivers was returned quickly and in much the same physical state, but warmer). Nearly all water used in HVHF is consumed,[24] and what is not is heavily polluted and must be treated before it is returned to the hydrological cycle. The long term effects of removing this water, effectively permanently, from the surface and usable groundwater reservoirs, therefore nominally from the hydrological cycle are as yet unknown. Withdrawals for HVHF should be monitored closely, because they are a systemic loss to the global hydrological cycle, unlike (most) withdrawals for other purposes.

This research is a tiny but important example of one of humanity's foundational assumptions: water must always be available where people live, or else people will not be able to survive. Pennsylvania's regulatory framework makes that clear by the near-absence of regulation surrounding water withdrawals. That is not unreasonable for the state—it has a network of large rivers and a relatively wet climate. However, water availability has always been assumed everywhere, not solely in wet areas, because until recently, the population of an area was determined in large part by water availability. Large engineering projects have always tried to change that geography, but their effects were mostly local and/or relatively small—ancient Egypt flourished by making use of the Nile River valley's fertility not through massive water transfers to outside the valley.

[24] Refer to footnotes on p. 14 for definition of consumptive water use.

Today, cities like Los Angeles and many others flourish because they are able to take water from elsewhere (Cousins and Newell 2015), without considering environmental and social costs because there was always been enough water. The modern world has the industrial capability to move massive volumes of water long distances, but not yet a way to understand and account for the costs and benefits (monetary and otherwise) of doing so. Political-industrial ecology can provide such an understanding.

A political-industrial approach to water resources could however be limited in its predictive capacity. Political ecology's interest in environmental subjects and degradation and marginalization requires the emergence of those things: if, for example, HVHF has never taken place in an area, there may be no environmental subjects coalesced around the issue, and no degradation and/or marginalization (from HVHF) to understand. On the other hand, industrial ecology's use of LCA is a highly useful modeling tool, but it is aspatial. Applying the results of such a study in a location different from where it was conducted may not yield wholly accurate results. However, such a risk may be worth taking. Political-industrial ecology should seek to be predictive where possible, but strike a balance with providing timely information. Some uncertainty may need to be accepted to meet that goal.

Chapter 6 Conclusion

Hydraulic fracturing, because of its vast (potential) extent and decentralized nature, is a new and important stressor on the water-energy nexus. To date, the technological process and its social and political influences have been well-researched, but they have only rarely been assessed within the same analytical framework. Understanding hydraulic fracturing as a political-industrial system operating at the water-energy nexus provides a path toward integrating those complementary viewpoints. Controlling but limiting water extraction for hydraulic fracturing allows society to meet its expressed need for hydrocarbon energy and ensure sufficient water availability for uses other than hydraulic fracturing.

The most important finding from this research is that Pennsylvania is well-suited to the development of hydraulic fracturing from the standpoint of water availability. With at least five major rivers partially or entirely in its borders (from east to west, the Delaware, Susquehanna, Allegheny, Monongahela, and Ohio Rivers) and many smaller (but still significant) tributaries, surface water is available in large volumes virtually everywhere and at most times throughout an average year. Some smaller cataloging units with high expected shale development, such as the Connoquessing Creek cataloging unit, may require several years' worth of discharge to provide sufficient water for HVHF while ensuring the riverine ecosystems remain healthy. However, many of Pennsylvania's highest expected density shale regions, such as the Upper Susquehanna-Tunkhannock cataloging unit, are also among the highest discharge cataloging units. Because cataloging units are layered, intense upstream extraction may have visible downstream consequences, as is the case in the Lower Susquehanna-Penns cataloging unit, among others. There, though there are no projected shale energy wells, intense upstream extraction could lower the discharge below environmentally-sustainable levels.

Due to the layered nature of cataloging units, the cascading effects of intense water extraction may be visible downstream. Pennsylvania law assumes the nearest shale well operator is responsible for downstream pollution or water diminution, but this phrasing is ambiguous. When thousands of wells are aggregated into one category—"upstream"—it may be virtually impossible to isolate the impact of one well from the cumulative impacts of all wells, thus leaving a potentially harmful environmental impact unmitigated. Such a real-world example of the MAUP presents hydrological and governance problems that are not easily analyzed or solved.

Attempts to minimize this problem require informed governance at the river basin scale, and the Susquehanna and Delaware River Basin Commissions are well-positioned. The two RBCs are also the most influential bodies in ensuring environmentally-sustainable levels of stream discharge are maintained. The SRBC, because it has by far the greater number of projected wells, is the primary organization for issuing permits for extracting water for hydraulic fracturing. These permits are highly specific in terms of maximum extraction rates per hour and per day, what the purpose of the extraction is, and the location. The permits (as well as this thesis) would benefit from increased focus on intra-basin water transfers in addition to the current interest in inter-basin transfers. An ideal permitting system would track water from where it was extracted to where it was used, not simply the purpose it will be used for as is the case now. Because water for HVHF is nearly entirely consumed, removing water from the cataloging unit it was extracted in represents a net loss to that cataloging unit and possible risk to the ecosystem. This could cumulatively be a large volume of water, but unless there is a change to the permitting process, there is no way to tell for sure.

A more complete understanding of Pennsylvania's surface water availability would also require many more water gauges, precisely sited. While using the cataloging unit scale is an improvement compared to the use of administrative scales, still smaller-scale analysis would be ideal. If gauging stations could be reliably placed at the confluence of all streams, the resulting data would enable still more accurate analysis of water availability. This knowledge could be applied to any water extraction, not just extraction for shale energy production.

Though much research has been done on the hydraulic fracturing process and its chemical, physical, and social mechanisms, little if any peer-reviewed research exists on the operational processes, and this is the primary source of assumptions for, and therefore limitations on, the accuracy of the modeling in this thesis. A lack of information on questions such as how long it takes to fracture a well, typical rates of water extraction, and any accepted, public effectiveness standards make it difficult to compare the process between places. Such information would have been useful for this research because it would have clarified exactly how rapidly water may be extracted and transported and how long it is usually stored for, among other questions. Without this information, it is difficult to model timelines (even idealized ones), and therefore determine long-term possibilities. Some or most of this information could be proprietary, and may be difficult to compare between firms and/or shale plays.

Further, because many shale basins are in arid areas around the world, it will not be possible to replicate the water abundant Pennsylvania experience everywhere. Hydraulic fracturing in such areas risks intensifying the existing hydrological vulnerability caused by existing stresses on water resources and the uncertain impacts of climate change. These factors should be carefully considered in the individual political, industrial, and ecological contexts. However, the method proposed in this thesis is broadly applicable if small adjustments are made. The most obvious adjustment would be the need to determine the water input requirements for each new shale play. As shown in Chapter 2, each shale play in the US requires different volumes of input water, and that is likely to be true for all geology across the world. Sufficient test wells must be created in order to build a base of initial data on volume of water required. Once that is established, a researcher using this method must account for any non-surface water sources used, particularly groundwater. Groundwater is the obvious choice to supplement surface water in areas with few lakes and streams, but the volume of an aquifer tends to be much more difficult to measure than that of a river, and aquifer recharge timelines vary widely. If groundwater use were authorized however, it could replace a portion of the volume that might be extracted from surface water sources. This would affect the linkages between ground and surface water which, though unexamined in this thesis, are hydrologically important and which change from place to place.

This research sought to conceive of the hydraulic fracturing system as a political-industrial ecology. Though the primary goal was to determine water availability for hydraulic fracturing, it should help to promote the use of political-industrial ecology as a valuable conceptual framework for the social sciences, especially those that seek to shape and understand the water-energy nexus. Hydraulic fracturing is an industrial system with major influence on society, but exists only in ways permitted by the societal framework that created it. Using water as a focal point allows an understanding of many of the most influential governance institutions. While these institutions may not govern the industrial process directly, by shaping the inputs needed by the process they shape the process itself, and through understanding this interrelationship we can gain a better understanding of the industrial process.

Hydraulic fracturing has been at the center of intense societal debate for a decade, and looks likely to remain so for the foreseeable future. Its social and environmental effects are highly contentious at many spatial and temporal scales. This research has shown a way to understand one part of the process. Future research should seek to deconstruct the process and its assumptions more fully. It may not be possible or desirable to eliminate water entirely from the system, though efforts are underway to do so. Space exists for research into many questions about hydraulic fracturing: what factors shape societal acceptance or resistance? What would make our current regulatory system better? Which societal processes are (re)made by HVHF, and are those processes beneficial or detrimental? And finally and most importantly, are the benefits of hydraulic fracturing worth the costs?

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