Environ Sci Pollut Res
DOI 10.1007/s11356-017-8970-0

REVIEW ARTICLE

Extensive review of shale gas environmental impacts from scientific literature (2010–2015)

Daniele Costa[1] **· João Jesus**[1] **· David Branco**[2] **· Anthony Danko**[1] **· António Fiúza**[1]

Received: 5 January 2017 / Accepted: 30 March 2017

Abstract Extensive reviews and meta-analyses are essential to summarize emerging developments in a specific field and offering information on the current trends in the scientific literature. Shale gas exploration and exploitation has been extensively debated in literature, but a comprehensive review of recent studies on the environmental impacts has yet to be carried out. Therefore, the goal of this article is to systematically examine scientific articles published between 2010 and 2015 and identify recent advances and existing data gaps. The examined articles were classified into six main categories (water resources, atmospheric emissions, land use, induced seismicity, occupational and public health and safety, and other impacts). These categories are analyzed separately to identify specific challenges, possibly existing consensus and data gaps yet remained in the literature.

Keywords Shale gas · Hydraulic fracturing · Fracking · Environmental impacts · Literature review

Daniele Costa and João Jesus both contributed equally to this work.

Responsible editor: Philippe Garrigues

Electronic supplementary material The online version of this article (doi:10.1007/s11356-017-8970-0) contains supplementary material, which is available to authorized users.

✉ António Fiúza
afiuza@fe.up.pt

[1] Centre for Natural Resources and Environment (CERENA), Department of Mining Engineering, Faculty of Engineering, University of Porto, Rua Dr. Roberto Frias s/n, 4200-465 Porto, Portugal

[2] Energy Planning Program, Graduate School of Engineering, Universidade Federal do Rio de Janeiro, Centro de Tecnologia, Bloco C, Sala 211, Cidade Universitária, Ilha do Fundão, Rio de Janeiro, RJ 21941-972, Brazil

Motivation and relevance

Shale gas exploration and exploitation remains shrouded in controversy. From a policy point of view, there are two conflicting perspectives: while some view the shale gas revolution as a step back on the reduction on fossil fuel reliance, others claim that shale gas can be regarded as a transitional fuel by substituting coal for electricity and heating. Furthermore, shale gas is also viewed as a way to decrease dependency on foreign sources of energy. However, doubts remain of the impact of shale gas exploration on climate change when its whole life cycle is considered (Howarth et al. 2011).

Controversy also arises from the point of view of its other environmental impacts and risks. While some find that the impacts and/or risks that shale gas exploration entails are unacceptably high and therefore should not be allowed under any circumstances, others believe that such impacts can be controlled and managed through a combination of reasonable and adequate regulation and risk assessments.

Regardless, it seems clear that initially observed environmental impacts were higher than a reflection of the infancy of a whole new industrial process, largely unregulated and unrefined at first. What remains uncertain, however, is whether recent developments and regulations (Cathles Iii et al. 2012; Howarth et al. 2012) were capable of sufficiently reducing or containing the negative impacts to acceptable levels. Therefore, the magnitude of both the environmental impacts and that of the novel procedures and regulations to reduce them are largely unknown.

To clarify these controversial aspects, a review of existing scientific literature is necessary. Although it may be considered that sufficient time has yet to pass for some environmental impacts to be noticeable, such reviews are important to identify existing consensus as well as identifying knowledge gaps, where research efforts should be focused. If

Published online: 28 April 2017

accompanied by risk assessment and modeling, such analysis, even if preliminary in nature, can serve as guidance to policy makers in the short to medium term, while further research is performed to increase the validity of identified consensus.

Recently, some authors have examined the growth of shale gas scientific and technical literature (Li et al. 2015; Prpich et al. 2016; Wang and Li 2016). For example, Lee and Sohn (2014) evaluated the state of technological development of shale gas in China and the USA by comparing the evolution of the number of patents over time. In addition, a bibliometric review by Prpich et al. (2016) focused on the environmental risk assessment for the requirements of UK regulators across the different production stages of shale gas exploration and exploitation, while Li et al. (2015) performed a generic bibliometric analysis of the scientific literature. Nevertheless, a systematic analysis of the existing (or lack thereof) consensus between different studies on shale gas environmental impacts as well as the impact of major mitigation strategies has yet to be made.

Considering the need to understand and identify what has been learned so far on the environmental impacts and risks, this article provides an extensive review of peer-reviewed publications in representative academic journals from 2010 to 2015 with the goal of examining the challenges and data gaps between research, current industry practices, and impacts of shale gas exploration and exploitation.

Methodology

The objective was initially to use a generic search to perform the widest possible search and allow for the identification of articles that assess shale gas and hydraulic fracturing from diverse perspectives. Therefore, scientific papers were obtained using SCOPUS using a simple search based on the terms "shale gas" and "hydraulic fracturing"; using the "or" operator in article title, abstract, or keywords for articles only; and considering the "and" operator for language equivalent to English. Articles missing key categories, such as the author's name or location, were excluded. Finally, a search of duplicates was conducted among the results obtained in each database.

Selection criteria, data collection, and assessment

Articles were evaluated covering the more recent 5 years of academic research from 2010 to 2015. Extending this to an earlier time frame was considered unnecessary due to the lack of studies available before that date. In 2007, shale only accounted for 8.72% of the total production of natural gas (NG) in the country (USEIA 2015). In addition, January 2007 was also the time when gross natural gases from shale formations were first reported by the US Energy Information Administration (USEIA). This assumption was further confirmed by a simple search between 2005 and 2010 where no relevant environmental impact assessment studies were found and recent review studies (Prpich et al. 2016).

Articles that discussed policies were only considered when they referred to environmental aspects and other impacts linked to shale gas. Similarly, discussions on energy security and shale gas extraction were excluded. Despite the fact that these other articles contribute to the discussion on shale gas development, they are considered outside the scope of this review, which focuses on the most relevant environmental impacts their management thus far.

In addition, studies focusing on the hydraulic fracturing process or stimulation technology, geology (such as fracture mapping, porosity modeling, among others), and wellbore integrity were not considered since these were also considered outside the scope of assessing environmental impacts.

Based on these criteria, the articles were classified as follows: (1) water resources, (2) atmospheric emissions, (3) land use, (4) induced seismicity, (5) occupational health and safety, and (6) other impacts. The areas covered in each of these six criteria are listed in Table 1.

Articles related to occupational and public health and safety were grouped based on exposure pathways (water or air) since most studies focused on either exposure to contaminated groundwater (for the general public), produced water and spills (for workers), or continuous exposure to air contaminants. Few published works, if any, report a combined exposure risk to these different pathways.

For the sake of simplicity, not all articles are necessarily referenced, specifically if the content is not particularly relevant, novel, or is limited in scope. After the articles were classified in one of the six impact categories, additional information for each article was also examined. These included the

Table 1 Article classification criteria according to impact categories

Impact category	Areas covered
Water resources	Groundwater and surface water contamination, depletion and water quality, and wastewater treatment
Atmospheric emissions	Air releases and quality, climate change, greenhouse gas emissions (GHGs)—including fugitive
Land use	Risk to biodiversity, noise impacts, increased traffic, waste management (including radionuclides)
Induced seismicity	Induced seismicity related to hydraulic fracturing and its practice
Occupational and public health and safety	Production accidents, spills, public health
Other impacts	Multiple impact evaluation, socioeconomic impact, synergetic impacts

geographic location of the article's corresponding author and also ranking the data source.

The geographic location of the articles was examined since it may be considered as a proxy to identify which are the most active locations of shale gas research, irrespective of different stages of development and implementation. This geographic location was classified according to the location of the first author from each paper and mostly reflects institutional interest/commitment to shale gas research. This approach is not without its limitation; due to increasing multi-national and multi-disciplinary collaborative studies, some data obtained through this method may not be representative, and thus, this approach should be analyzed with care and seen as preliminary.

The geographic groups were then classified as follows: (1) USA, (2) Canada, (3) UK, (4) China, (5) Europe (including Russia but not the UK), and (6) others, which included articles that did not belong to any of the other five groups.

Articles were also ranked according to the data source, as suggested in similar studies (Prpich et al. 2016). This includes (1) primary data sources, (2) secondary data sources, and (3) theoretical studies. Descriptions of each of the three ranking systems used are as follows:

- Primary data sources are those articles that collected, provided, or evaluated direct measurements or field data. This provides new information on impacts caused by hydraulic fracturing from shale gas extraction. This includes laboratory experiments, modeling studies, or even surveys.

- Secondary data research are those articles that offered reviews on shale gas production, but did not offer new data and only systematically discussed impacts caused by shale gas exploration and exploitation. Studies in this group provided critical reviews of the literature and have the potential to support policies and best practices for shale gas production.

- Finally, theoretical studies are those that adopted a mixed method approach, where a qualitative or quantitative evaluation of the topic was done with non-empirical data to support the assessment of impacts and risks. Both case studies and studies that evaluated or used raw data as a reference from third parties were classified here.

After the classification of all articles, a detailed analysis of the major environmental impact categories shall be presented in this study to assess existing consensus and major research data gaps divided in the following categories: water resources, atmospheric emissions (air quality and climate change), land use, induced seismicity, and multiple environmental impact assessment (life cycle assessment (LCA) and other studies).

Results and discussion

In total, 3882 articles were identified based on the initial search parameters, of which 701 were identified as suitable for understanding environmental impacts. Out of these, 373 were not accessible or unavailable and were not included in this review. This left 328 articles that were included in the evaluation and were classified according to the six impacts defined in the "Methodology" section for each year from 2010 to 2015, and the result of this classification is shown in Fig. 1.

It is important to note that no articles were identified or obtained that fit the criteria prior to 2010, and therefore are not represented in the Fig. 1. Reports on NG production from shale formations by the USEIA (2015) only began in 2007, which explains the lack of articles that fit the criteria for the search between 2005 and 2010.

The growth of the number of articles during this time may reflect the production of NG in the USA, which grew an average of 43% between 2007 and 2011 and lead to a reduction of annual NG prices from 7.97 USD in 2008 to 2.66 USD in 2012 (USEIA 2015). The combination of increased production and lower prices changed the North American energy market. In addition, the identification of technically recoverable shale gas reserves in other parts of the world leads to a debate on the viability of this technology to reduce the dependency on energy imports, particularly in Europe (USEIA 2013). Additionally, it may also represent a concomitant increase of public and scientific awareness of shale gas exploration and its potential impacts and risks.

An examination of Fig. 1 further demonstrates a significant increase in number of shale gas articles between 2010 and 2015, from 2 in 2010 to 121 by 2015. Varying proportions were observed for four of the five topics during this time frame, namely, induced seismicity, land occupation, health and safety, and atmospheric emissions. Health and safety always showed the least percentage out of all six classifications and varied between 0 and 32%, and there was a steady increase in the percentage of articles for water resources, from 0% in 2010 to 50% by 2015.

As the number of shale gas articles increased, so did the geographic coverage. As seen in Fig. 2, only two regions were represented in 2010 ("USA" and "others"). Afterwards, five regions were included by 2012, and all six were represented by 2013 and continued that way for 2014 and 2015. It should be noted that even though Argentina is currently one of the

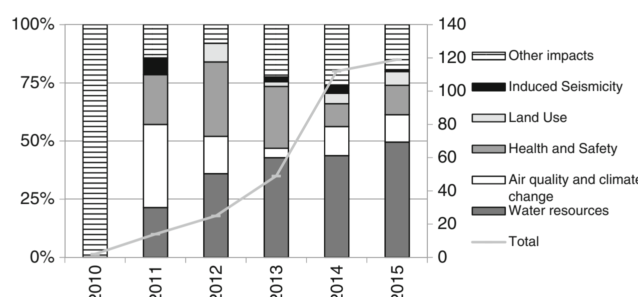
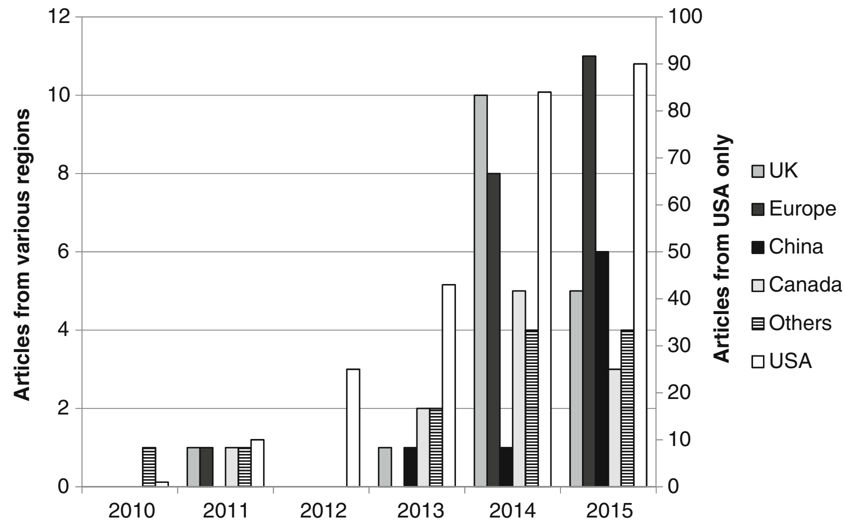


Fig. 1 Evolution of included articles per impact category

Fig. 2 Geographic coverage of included articles—USA in secondary axis



few countries commercially producing shale gas, no articles from this country were found.

Several regions saw significant increases in 1 year—from 2013 to 2014—Canada, UK, Europe, and others—and from 2014 to 2015 for China. Even though this is the case, the majority of shale gas articles from the studied time frame was from the USA (around 79% of the articles), followed by Europe (6%) and the UK (5%). Cooperation among universities across countries is still low and accounted for only 39 articles, with approximately 77% of mixed nationalities involving the USA.

Water resources

This section discusses recent developments and current practices of water management in shale gas exploration and exploitation, including spills, water usage, and treatment from the 141 included articles. Of these, primary data (type 1) represented 33% in 2011 and a maximum of 81% in 2013, while secondary data (type 2) peaked in 2011 at 67% but ranged between 7 and 22% in the remaining years. Finally, theoretical studies (type 3) ranged between 10 and 44% of total number of papers in water resource category. Out of these 141 articles, only 17 were not from the USA with 7 from different European countries and 3 were from the UK and were published only in 2014 and 2015.

The hydraulic fracture water cycle can be described as having the following stages: (1) water acquisition, (2) chemical mixing, (3) well injection, (4) flowback and produced waters (wastewater), and (5) wastewater treatment and waste disposal (USEPA 2011; USEPA 2015a). Water contamination issues associated with shale gas extraction are usually associated with the contamination of surface water, treatment and disposal of produced water, and water management issues due to conflicting uses.

USEPA (2010) reports water use of up to nearly 19 m^3 per well, depending on its condition (depth, horizontal distance, and geologic factors), the number of times the well is fractured, and type of fracturing fluids used. Therefore, typical values vary significantly for each shale play (GWPC 2009). Discrepancies over the amount of water used in hydraulic fracturing are also found in different sources (Abdalla and Drohan 2010; Chang et al. 2014). From a life cycle perspective, Clark et al. (2013) demonstrated that water consumption per energy generated is different for each shale gas play evaluated. Nevertheless, it is always higher when compared to conventional gas produced in the same country.

Nearby water resources may come under pressure since hydraulic fracturing involves the pumping of large volumes of water into shale formations. This increased use in water resources may cause decreases in base flow to streams (Nicot and Scanlon 2012), changes to the aquatic ecology (Gallegos et al. 2015), and conflicts with other industries that use this water, such as agriculture (Goodwin 2014).

Therefore, the industry is examining ways to decrease their water requirements by reducing water intensity per well in shale gas explorations. However, increasing horizontal well length may lead to increasing water consumption per well (Nicot et al. 2014). It is important to note that net water use for shale gas exploration and exploitation was found to be within the range of other energy sources, namely, coal (Goodwin 2014; Nicot and Scanlon 2012) and uranium mining (Nicot and Scanlon 2012). Although cumulative water consumption may result in extra pressure on water resources since demand rises, these impacts are regional (Jackson et al. 2015) and basin specific (Pacsi et al. 2014).

One alternative option of water for drillers is to use municipal or tap water, which do not require extensive pretreatment prior to use in shale gas operations. These water sources accounted for 29% of hydraulic fracturing water in parts of Pennsylvania (Abdalla and Drohan 2010). Acid mine drainage (AMD) is another alternative water source for drillers in regions such as the Marcellus and Utica regions. This reduces freshwater demand but typically requires water treatment prior to its use for hydraulic fracturing (Abdalla and Drohan 2010; Rodriguez and Soeder 2015). Seawater and brine groundwater have also been successfully used in both onshore and offshore hydraulic fracturing (Rodriguez and Soeder 2015). Both of these may be options for onshore projects in arid regions or in areas with water scarcity.

Shale gas wastewater—contaminants and sources

Wastewater derived from shale gas exploration and exploitation may be classified into three main types, based on different processes as well as different operational periods. The first type is drilling fluids. As the name suggests, it is wastewater resulting from the initial drilling of the well before any hydraulic fracturing or gas extraction can occur and it is normally used to cool and lubricate the drill bit and clean drilling cuttings (Lutz et al. 2013).

The second type is the flowback fluid. This represents the initial flow of wastewater immediately after hydraulic fracturing, and it resembles the fracturing fluid particularly because it contains organic compounds, even though it is a mixture of fracturing fluid and native existing fluids. It is estimated that 10 to 40% of the water injected into a well are returned to the surface as flowback water. Flowback fluid mostly occurs in the first 7 to 10 days but can be up to 4 weeks after hydraulic fracturing (Barbot et al. 2013; Haluszczak et al. 2013). It may represent 32.3% on average of the wastewater volume produced during the life span of a well (Lutz et al. 2013). Other names used to describe this wastewater type include flowback brine and fracturing water flowback.

Finally, the third type is water produced. This comes from the recovery of naturally occurring fluid from the shale formation itself mixed with a small volume of fracturing fluid and flows through the entire life span of the gas well. Although it should be mentioned here that there is no standard definition of flowback fluid and produced water, they are often grouped together and the distinction between the two is difficult to make in many instances. Because of this, other authors have suggested the use of an additional term (transitional water) to distinguish between the two different phases (Bai et al. 2015).

The composition of flowback and produced water may vary significantly. Organic compounds that can be found in both flowback and produced water include surfactants (Thurman et al. 2014), low levels of volatile and semivolatile organic compounds (volatile organic compound (VOC) and SVOC) (Akob et al. 2015; Lester et al. 2015; Shih et al. 2015; Ziemkiewicz and Thomas He 2015), low levels of polycyclic aromatic hydrocarbons (PAHs) and other aromatics (Maguire-Boyle and Barron 2014), and high values of low molecular weight alkanes and alkenes and total organic carbon (TOC). An important aspect is the potential creation of halogenated and non-halogenated compounds as a consequence of the reactions between the fracking fluid and the rock matrix (Maguire-Boyle and Barron 2014).

Naturally occurring radioactive materials (NORMs) may also be found both in produced and flowback waters (Alley et al. 2011; Gregory et al. 2011). Although a recent study mentioned that NORM concentrations may be higher in produced water (Shih et al. 2015), the NORM found in these wastewaters may be dependent on the type of rock formation. Non-radioactive cations and anions (salts) also depend on rock formation, similar to NORM. However, in this case, other researchers mention that rock formation may not completely explain salt concentrations in early flowback fluids and concluded that unknown reactions between flowback and the source material lead to increasing cation concentrations (Barbot et al. 2013). Therefore, inorganics in early flowback waters may not be a result of mobilizing compounds that naturally occur within the rock matrix.

In contrast to that, cation and anion concentrations in late flowback and produced waters may be explained by simple dilution of the existing brine formations with the fracturing liquid rather than from the introduction of these compounds from the fracking fluid itself. This statement is based on the fact that the same conclusion was reached using independent samples in the Marcellus shale play from two different research groups and institutions within the same state (Pennsylvania) and with no author overlap (Barbot et al. 2013; Haluszczak et al. 2013). Although, it is still unclear whether these results apply only to the Marcellus shale gas plays or to other shale regions in the USA.

The above studies indicate that fracturing additives as well as the fracturing process have a small contribution to inorganics in these wastewaters. In fact, other researchers suggest that fracturing additives may only make a small contribution, not only to inorganic compounds but also to organics and NORMs in flowback and produced waters (Ziemkiewicz and Thomas He 2015). Although, it should be noted that organic compounds are more likely linked to fracking fluids in these cases (Akob et al. 2015; Orem et al. 2014).

Comparisons may be made between shale gas produced water to other sources of NG in order to provide context. Maguire-Boyle and Barron (2014) compared shale gas with coalbed methane (CBM)-produced waters. Shale gas wastewater has a significantly higher TOC than CBM and slightly higher aliphatics but lower PAH and aromatics. As such, this may potentially mean that the water produced by shale gas is less toxic and more biodegradable in certain instances. Comparing with conventional gas, Pancras et al. (2015) reported higher lithium, potassium, and boron values for shale gas-produced water but lower copper and aluminum within the same gas region. Although, this similarity between conventional and shale gas-produced water may only be limited to inorganic substances such as salts and heavy metals.

Contamination may not only be caused by the introduction and extraction of fracking fluids into the subsurface but may also be a result of accidental spills or flaws in well construction. Recently, EPA published results of a systematic review of spills related to shale gas across 10 states in the USA from 2006 to 2011 (USEPA 2015b). From the 36,000 spills identified within the selected states, 33% could not be associated with hydraulic fracturing and only less than 1.3% (457 spills) were related to hydraulic fracturing.

Of that, flowback and produced water comprised 50%, while 20% were from the fracturing fluid. In addition, almost half of the total number of spills (46%) originated from storage and were mostly caused by human error. Also, the majority of releases were of a relatively small volume (13 m^3 or less) compared to the total amount of fluid used in hydraulic fracturing. Although, it is important to note that the number of spills increased three times from 2006 to 2011 and that approximately 70% of the spilled material were not recovered, arising 23% from unidentified sources (for example, which individual well or wells caused the contamination).

Other authors also addressed issues associated with spills. For example, during 2008 and 2013, Brantley et al. (2014) that reported that 32 spills (with a minimum volume of at least 1.5 m^3) originated from only 20 wells during a period when 6000 wells were drilled and 4000 were complete. Another study suggested that different processes in well drilling (the use of multi-well pad versus a single-well pad) lead to fewer environmental spills per well (Manda et al. 2014).

Another source of contamination may be from the migration of methane and salts to groundwater as a result of the fractures that were made during the fracking process (Heilweil et al. 2015; Jackson et al. 2013; Osborn et al. 2011). Although, this may not happen at every site since other studies have not shown any evidence of significant migration (Kolesar Kohl et al. 2014; Molofsky et al. 2011; Nelson et al. 2015; Warner et al. 2012; Warner et al. 2013b) and still others reported inconclusive results (Alawattegama et al. 2015; Hildenbrand et al. 2015).

In addition to the abovementioned contamination processes, poor treatment of wastewater (at public centralized treatment plants) may lead to the discharge of untreated contaminants into surface water bodies (Bowen et al. 2015; Getzinger et al. 2015; Kassotis et al. 2014; Lutz et al. 2013; Pancras et al. 2015; Skalak et al. 2014; Warner et al. 2013a). These treatment processes will be addressed in the next section.

Wastewater treatment

Disposal of flowback and produced water is of particular concern because of their volume, high salinity, and the presence of other compounds, such as organics, inorganics, and NORM, due to their ecotoxicological impacts. The main disposal methods reported in the literature include deep well injection, municipal wastewater treatment plants, and use as a deicing agent (due to the high salt content), among others (Maloney and Yoxtheimer 2012).

Deep well injection is the final destination of up to 95% of produced wastewater from conventional and unconventional onshore NG exploration (Lutz et al. 2013). However, this option may not be available in all the areas due to geological (for example, the Marcellus play) or infrastructure limitations. In this case, wastewater is sometimes transported to regions where deep well injection is available or sent to other treatment systems, such as municipal wastewater treatment plants.

Deep well injection may also be unavailable due to legal restrictions; in the USA, for instance, North Carolina banned deep well injection (Adair et al. 2012), while West Virginia and Pennsylvania (with only three and seven disposal wells, respectively) highly restricted this practice (Lutz et al. 2013). In Europe, different interpretations of the EU water framework directive have led to country or regionally specific bans all over Europe (Elsner and Hoelzer 2016).

As an alternative, it was common to dispose wastewaters to be treated at municipal treatment plants. However, treatment provided by these facilities was impaired since they are designed to treat domestic wastewater and are not prepared to treat high salinity levels (USGPO 2016). This incompletely treated water was discharged and impacted surface waters (Mauter and Palmer 2014). As a result, the practice was formally banned by the USEPA and pretreatment standards were established under the Clean Water Act for wastewater discharges to municipal treatment plants from onshore unconventional oil and gas (USGPO 2016). These pretreatments standards mainly focus on zero discharge to public-owned treatment works (POTWs) and surface waters by diverting the wastewater mainly to deep well injection (where available) or centralized waste treatment (CWT) facilities treating other industrial wastes. However, it remains unclear how many of these CWTs are capable of significantly reducing certain types of contamination, namely, the high inorganic salt content. In addition, CWTs are capital intensive and require a large number of wells to be cost-effective (Gómez et al. 2015). Construction of these CWTs is already a limiting factor in shale gas expansion at many locations in the USA.

Since traditional wastewater treatment methods have a limited capacity to treat these waste streams and deep well injection may not be an option, other methods have been suggested in the literature but only a select few have been used. These include microbial mats (Akyon et al. 2015); electrocoagulation (Ferrer and Thurman 2015); oil/water separation. ion exchange, freeze-thaw evaporation, thermal distillation coupled with crystallization, constructed wetlands, and reuse for irrigation (Gregory et al. 2011); advanced oxidation (Lee et al. 2015); microfiltration and ultrafiltration (He et al.

2014); and reverse or forward osmosis (Hickenbottom et al. 2013).

Many of the above treatment options have a marked limitation that restricts their applicability in the field. For example, reuse in irrigation or treatment using constructed wetlands is severely limited by the plant salt tolerance to such high levels of salinity, which are often higher than seawater. Freeze-thaw and thermal distillation are best applied in specific climatic conditions. Most remaining treatments are limited by very high costs and are energy intensive, such as reverse osmosis.

New technological developments are needed for more cost-effective treatments in order to provide valid options when deep well injection is not available. This is particularly true for salt removal due to the large volumes of wastewater. This also applies to the Marcellus play, where deep well injection is extremely limited. In addition, the climatic conditions there preclude the use of thermal distillation and evaporation as a treatment option. Another potential option may be the use of forward osmosis since it has been more extensively studied in recent years (Hickenbottom et al. 2013). This is because it may reduce costs when compared to reverse osmosis. Although, there is no evidence yet of the application of this technology in the field for this wastewater.

Produced wastewater also contains organics, which may be very diverse and complex and potentially difficult to treat. However, several articles refer to the high biodegradability of the waste stream, which is potentially due to high BOD/COD ratios and the high concentration of simple aliphatics (Kekacs et al. 2015; Lester et al. 2015). Concerning toxicity, control tests were more toxic than the raw hydraulic fracturing fluid and produced water in an acute toxicity test (Microtox) with *Vibrio fischeri* (Steliga et al. 2015). This suggests that the added chemicals in the hydraulic fracturing process are not toxic to living organisms. However, the compositional variability and chemical complexity of the added organic compounds, as well as lack of disclosure of fracturing fluid composition (Kekacs et al. 2015), hinders researchers' ability to assess both the biodegradability and the toxicity. As a result, further tests examining both chronic and acute toxicity appear to be warranted.

Finally, wastewater reuse is yet another wastewater management option and may be applied directly or following dilution or pretreatment. However, it may be limited by the chemical stability of viscosity modifiers and salt precipitation due to barium and calcium (Haghshenas and Nasr-El-Din 2014). A series of simple pretreatment steps may enhance reuse by precipitating most of these salts or controlling pH. Eventually, treatment is no longer effective in removing these cations and reuse becomes unfeasible.

Based on the information examined in the articles, the preferred management strategy may be a compromise between water quality, economic constraints, and process performance. This suggests that one option may be to pretreat produced water followed by reuse. This may be done in conjunction with blending with makeup water and finally followed by deep well injection (where available).

Atmospheric emissions

This section discusses recent developments in monitoring of air quality and GHG and exploitation and impacts on health. From the total articles selected, 39 evaluated atmospheric emissions. Initially, more interest was shown in this topic, as 36% of suitable articles focused on this theme in 2011. Although, there has been a steady decline in the percent overall contribution since then.

Research in this topic has predominantly been carried out in the USA (79% of articles tracked) over the examined time frame, but there has been an increased number of articles in 2014 and 2015 from "Europe" and the "UK." In addition, only type 3 articles were identified in this area and totaled 65% on average. Finally, it is important to note that the last 2 years contributed to 72% of the total number of articles in this impact category from 2010 to 2015.

Air quality

The fast development of shale gas in proximity to residential areas and heavily populated areas has raised concerns on the impact of local and regional air quality. Although, there remains a lot of uncertainty over this issue to date. This may be related to the fact that air pollution generated by the shale gas industry is extremely difficult and costly to monitor. For example, sampling must take place over a long period of time in order to obtain robust results. Therefore, it is not surprising that only a small number of suitable articles (9) was found with reports of raw data emissions. In addition, no articles were found from "Europe" that looked at this issue, but this may be related to the very limited shale gas activity compared to other regions.

Of the articles that were published, comparisons between the studies were limited due to the extremely heterogeneous nature of the data collected, number of samples taken, the type, and even the specific compounds that were analyzed, among others. Nevertheless, some general trends were found through the analysis of all three types of suitable articles.

Emissions are generally classified into the following categories: VOCs, PAHs, particulate matter (PM_x), NO_x, SO_x, carbonyls—such as formaldehyde (Colborn et al. 2014), and ozone, a secondary pollutant resulting from the reaction of NO_x and VOC in the presence of solar radiation (Ahmadi and John 2015; Edwards et al. 2014; Swarthout et al. 2015). One important contaminant that was only addressed in one article was radon. Walter et al. (2012) examined this issue from drill cuttings. Although, emissions from other waste materials (both solid and liquid) generated from shale gas exploration and exploitation have not yet been addressed in the literature.

There is a great variety of equipment that may be considered a source of air pollution either through combustion or fugitive emissions. For combustion, an assortment of equipment (generators, compressors, among others) utilize diesel engines during their operations, since they are traditionally used in shale gas exploration and operational activities and emit a variety of the air pollutants listed above (Rutter et al. 2015).

Litovitz et al. (2013) and Ethridge et al. (2015) inventoried combustion and fugitive air emissions through a survey of various entities producing in the Barnett shale area. This included produced water storage tanks, piping component fugitive areas, blowdown vents, condensate storage tanks, engines, process vents, oil storage tanks, and heaters/blowers. The results showed that combustion emissions encompassed less than 10% of emissions, while emissions from storage tanks, vents, and piping summed to almost 80% with 50% coming from just produced water storage tanks and piping. Heaters and boilers emitted the least (1.3%) (Ethridge et al. 2015). Additional studies are needed from the Barnett and other plays to determine if similar results are obtained.

Emissions of air contaminants occur during various phases of shale gas exploration and exploitation, including initial drilling, hydraulic fracturing, well completion, and production operation. A recent study concluded that emission standards would not be exceeded in Poland during exploration activities despite the high level of NO_2 emissions (Bogacki and MacUda 2014). Colborn et al. (2014) determined that emissions were higher during initial drilling.

Litovitz et al. (2013) estimated that well site preparation may emit between 150 and 170 kg VOCs, 3800–4600 kg NO_x, 87–130 kg $PM_{2.5}$, 87–130 kg PM_{10}, and 3.8–110 kg SO_x and 46–1200 kg VOCs, 520–660 NO_x, 9.9–50 kg $PM_{2.5}$ and PM_{10}, and 3.1–4 kg SO_x per well during production. Although these values are estimates, it is important to highlight that emissions for site preparation values tend to be higher in NO_x and SO_x due to the influx of traffic to the facilities.

Emissions may also vary depending on seasonal effects, particularly for ozone formation (Edwards et al. 2014) and the shale play in question. For example, lower concentrations of VOCs were found for the Marcellus shale compared to the Barnett shale play (Goetz et al. 2015). Although, the authors noted that the results of air quality studies should be examined on a case-by-case basis and that caution should be used in generalizing the results. Finally, much like other fuels, the impact of shale gas on air quality can be significant. Although, it is important to note some studies. (More recently, Song et al. (2015) indicated that overall emissions remain lower than those of coal.) This suggests that the commonly used policy of shale gas as a transitional fuel from coal should continue to play a part.

An important aspect with air pollution is that contaminants might be native to the shale basin that is being explored or exploited. For example, a recent study concluded that secondary organic aerosols from sources unrelated to oil and gas development were the cause of ozone formation (Rutter et al. 2015). In addition, emissions are not exclusive to unconventional shale gas exploration and exploitation or a direct result of the fracturing process. This is especially true in areas where conventional gas exploitation is also occurring. Therefore, it is important to obtain air quality measurements prior to the exploration and exploitation of shale gas in order to delineate the contribution of this activity to background air quality.

Public health risks to surrounding communities are still a controversial issue. Bunch et al. (2014) indicated that VOC levels due to fracking activities did not pose excessive exposure risks to their communities. Although, another study (McCawley 2015) showed a link between respiratory effects from air contaminants to both the shale gas extraction itself and the heavy traffic associated during construction and exploration activities. This is due to emissions not only from PMs, VOCs, and PAHs but also from crystalline silica (McCawley 2015). One study from the UK focused on inhalation of hydrocarbons from operational air emissions over the lifetime of a well and estimated increased health risks due to this exposure (Reap 2015).

There are less studies on the impact to workers. Recently, OSHA and NIOSH (2015) have reported that workers involved in hydraulic fracturing activities are exposed to dust with high levels of breathable crystalline silica. Rosenman (2014) also examined this issue and estimated appreciable risks after long-term exposure.

Other studies examined exposure of both workers and communities with differing results. Several studies mentioned low to no substantial risks of exposure for both of these groups (Bunch et al. 2014; Ethridge et al. 2015; Goetz et al. 2015). Nevertheless, Paulik et al. (2015) and Colborn et al. (2014) alerted to potential dangers. The different conclusions of these studies may be a result of monitoring different compounds. For example, Paulik et al. (2015) focused on only exposure to PAHs. In addition, it is important to note that permissible levels may not necessarily take into account segments of the population at higher risk of adverse health effects such as pregnant women and infants (Colborn et al. 2014).

Considering these potential risks, additional research efforts are needed since long-term direct measurements of air pollutants are extremely scarce (Goetz et al. 2015; Roy et al. 2014). This is especially true to obtain data for multiple years coming from different shale plays and regions while monitoring for the contaminants listed earlier in this section, especially radon. As such, data from these new studies would provide the basis for potential mitigation measures as well as the risk assessment of air pollutants to workers and public health in general. If measures are needed, two different strategies may be used (alone or in combination) for the protection of human health: first, the potential decrease of exposure to pollutants either by best practices or mandated regulations or second, the reduction of the pollutant load through technical improvements and mitigation strategies.

The first option would be the enactment of new regulations. Several regulatory measures have already been suggested in the literature with some of them already implemented. For example, at least 20 states in the USA have established setback requirements regulating the distance between exploratory areas and residential areas at a range from 300 to 3000 m (Richardson et al. 2013). Other proposed that regulatory changes may include proposals to aggregate industry sources and the requirement to use best available technologies (BATs) (Litovitz et al. 2013).

The second option would be the use of alternative chemicals and technologies that focus on limiting fugitive emission (Centner and Petetin 2015). For example, one option to consider would be the implementation of dual-fuel technologies, such as those that operate with diesel and NG (Thorn 2015). Others include the use of complete combustion devices to reduce VOC emissions, incineration of aromatics and heavy hydrocarbons, the use of high-bleed controllers (Centner and Petetin 2015), or the application of selective catalytic reduction for NO_x emissions and diesel particulate filters for $PM_{2.5}$ (Roy et al. 2014).

Climate change

The climate change section will focus on the two main direct GHG resulting from shale gas exploration and exploitation, namely, methane and carbon dioxide. However, measurements and/or estimates of these emissions are difficult to assess directly in the field due to a wide array of technical difficulties that can be condensed in three reasons. First, direct measurements of methane emissions are scarce and differ significantly. For example, Allen et al. (2013) reported emissions from well completions to be 98% lower than the national estimates by USEPA. This discrepancy may not only be due to differences in methane source allocation but also to restricted access to random sampling locations since those selected may have been potentially chosen by industry since they may have been the best performing options (Howarth 2014).

Second, methane leakage rate is an extremely important value for GHG estimations, though widely contested in the literature. Simply defined as the percentage of methane leaked over the total NG produced, methane leakage rate estimates vary from 0.42% (Allen et al. 2013) to ranges of 0.66–3.9% (Jiang et al. 2011) and even as high as 3.6–7.9% (Howarth et al. 2011). Furthermore, these estimates are likely to be play specific (Peischl et al. 2015) and dependent on final well life span (Howarth et al. 2012). Some reported values are contested as either being too low (0.42% indicated by Allen et al. (2013)) or too high (the upper limit of 7.9% indicated by Howarth et al. (2011)). Third, an aspect that remains poorly discussed in the literature is the possibility of refracturing existing wells and their impact on GHG emissions (Jiang et al. 2011; Stephenson et al. 2011).

Although all of the issues listed above are extremely important, they represent only part of the total GHG emissions in the life span of a shale gas well. For the evaluation of total GHG emissions, LCAs are often performed for more accurate assessments (Burnham et al. 2012; Howarth et al. 2011; Jaramillo et al. 2007; Jiang et al. 2011).

Heath et al. (2014) developed a systematic review of eight LCA and concluded that emissions from shale gas averaged approximately 488 CO_2 equivalent/kWh. However, LCA also have significant variations in the chosen parameters, which are highly debated among authors in the reviewed literature. These parameters include GHG time frame (Cathles Iii et al. 2012; Howarth et al. 2011; Howarth et al. 2012), the end use of the produced shale gas, and the considered methane leakage rate (as discussed above). These discrepancies limit not only an accurate assessment of total GHG emissions over the life cycle but also comparisons with other energy sources, such as coal.

It is important to note that different end uses (heating or electricity production) involve different considerations and potentially impact different input parameters and output results (Cathles Iii et al. 2012; Howarth et al. 2012). For example, Howarth et al. (2011) concludes that shale gas GHG emissions are higher than coal for heating while other studies suggest that shale gas is substantially better than coal with 38–50% less GHG emissions if electricity production is considered (Chang et al. 2015; Jiang et al. 2011; Stephenson et al. 2011). Similarly, conflicting results were also reported for conventional versus shale gas operations for GHG emissions. Heath et al. (2014) concluded similar emissions for this energy source, while other authors report an increase of 1.8 to 17% for shale gas over conventional gas (Jiang et al. 2011; Stephenson et al. 2011).

An important aspect that may impact and change the values obtained in these LCA's are the proposed or implemented mitigation strategies in order to attenuate total GHG emissions. This focus has primarily been on initial well completion, since methane leakage may be extremely high during this process. In order to mitigate these GHG emissions, a wide variety of technologies are available and are referred as reduced emission completions (RECs) (Cathles Iii et al. 2012; O'Sullivan and Paltsev 2012; Stephenson et al. 2011).

One alternative option to venting is to recapture with the intention to sell. This option may be economically feasible considering that expected methane losses are much higher during well completion of shale gas than conventional gas because of hydraulic fracturing (O'Sullivan and Paltsev

2012). From a regulatory standpoint, the USEPA defined in 2012 that each well completion occurring after January 1, 2015 must employ REC in combination with a completion combustion device (flaring) (USEPA 2016).

Other technologies that may be considered are carbon capture and storage (CCS) in depleted shale gas reservoirs and the use of supercritical CO_2 as a working fluid in hydraulic fracturing. However, studies on CCS in depleted shale gas reservoirs (Wang et al. 2011) have yet to prove that the sequestration capacity is sufficient to offset overall GHG emissions from the industry (Edwards et al. 2015). Supercritical CO_2 has the potential to simultaneously reduce water requirements and sequester CO_2, thereby reducing two critical aspects of shale gas production (Middleton et al. 2015; Wang et al. 2012). However, additional tests are needed to determine the efficacy of this technology in the field.

Land use

Land use can be defined as the conversion of land from one type of biome/management to another (IPCC 2014). This impact category shows a wide range of impacts as demonstrated in the 15 examined articles from 2010 to 2015. This number represents approximately 5% of the total suitable articles in shale gas impacts. This classification is predominantly constituted by type 1 articles (40% on average) and became more representative in 2014. The geographic locations were only from the USA, Canada, and the UK.

Shale gas exploration and exploitation involves various building activities in the selected area. Following the successful identification of potential areas using different methodologies, well pad construction requires not only the removal of soil and vegetation but also the transport, handling, and storage of chemicals and other materials for the building of gas pipelines, water extraction structures, and other operational facilities. All of these activities are liable to impact land use and cause habitat disruption, erosion, and increase noise pollution (Drohan et al. 2012; Moran et al. 2015; Olmstead et al. 2013). Finally, road improvements may be required in order to handle the increased traffic during this phase, although this increased volume may potentially increase traffic accidents in the play area (Graham et al. 2015).

Land use and area occupied by shale gas is highly dependent on a variety of factors, including the number of wells per pad, well pad size, and distance between them. While a larger number of wells per pad allow for less direct land coverage as support infrastructures are more concentrated, it also means wider spacing between well pads. This may impact pipelines and road construction needs as well as it intensifies the potential environmental impacts locally (Baranzelli et al. 2015).

The average building area for the different components involved in shale gas exploration and exploitation varied in the analyzed literature. The actual building area for well pads have been reported or assumed to be between 1.2 and 3.55 ha for well pad with two or less wells (Baranzelli et al. 2015; Moran et al. 2015) and between 2 and 9.93 ha for well pads with 8 to 16 wells (Baranzelli et al. 2015; Racicot et al. 2014). If adjacent infrastructures (compressor stations, storage areas for water, wastewater, and chemicals) are included, then the total building area varied between 3.56 and 13.68 ha (Baranzelli et al. 2015; Kiviat 2013).

Spacing between wells is also important in terms of proper land use allocation. This value is dependent on both legal requirements and technical issues of gas recovery when extracting from horizontal wells. Other authors report spacing requirements between 32 and 1024 ha for 2 and 16 wells per pad, respectively (Baranzelli et al. 2015). This spacing may also impact pipeline and road needs. Studies have reported average lengths per well between 2.3 and 2.8 km of pipeline (Evans and Kiesecker 2014; Racicot et al. 2014) and 0.73 km of road (Racicot et al. 2014).

While all the aforementioned parameters may be reasonably estimated based on the observed density of already explored or exploited areas, indirect land use changes are far more complex to evaluate. In addition, this indirect land use is often difficult to measure as shale gas exploration and exploitation may also impact overall land use due to associated industries (Moran et al. 2015). All of this, results in values that are much more variable compared to other aspects. For example, Moran et al. (2015) reported that 0.5 ha of natural forest was affected per well, while Evans and Kiesecker (2014) and Kiviat (2013) reported values of 8.6 ha of indirect land use impacted and 15 ha of affected forest per well, respectively.

The resulting impact of shale gas exploration and exploitation construction activities mainly result in risks to biodiversity due to direct impact on habitat fragmentation and pollutant dispersion. These risks are still poorly investigated in literature, which may be due to the required time to observe this type of impacts.

Six articles evaluating damages to ecosystems were identified in this review and pointed to the fact that many of the impacts caused by hydraulic fracturing were related to the poor management of chemicals, spills, or the improper handle of flowback and produced waters and other materials (Kiviat 2013; Latta et al. 2015). Shank and Stauffer (2015) and Latta et al. (2015) found similar results but focused on negative impacts to biodiversity. These studies showed that shale exploration led to reduced biodiversity and bioaccumulation of heavy metals in aquatic organisms and birds.

However, data on biodiversity impacts may be conflicting. For example, Shank and Stauffer (2015) found limited impacts on macroinvertebrate and fish, while Stearman et al. (2014) did not find any relationship between analyzed species abundance and shale gas exploration and exploitation. Some reasons to explain the seemingly lack of relevant impacts on ecological systems are the effectiveness of protective measures but more importantly the lack of sufficient time to observe these impacts (Shank and Stauffer 2015). This suggests that future research on ecological impacts is needed to truly assess the cumulative impact of shale gas over the entire life cycle of production.

Another aspect of land use relates to waste management and disposal. Mykowska et al. (2015) determined that the examined wastes have an estimated absorbed radiological dose lower than the average amount for individuals. However, previous studies with conventional oil producing site wastes suggest that NORM (including radium) may be present in produced sludges (Garner et al. 2015). The differences observed in potential risk between these two studies may reflect geological conditions in the different analyzed basins. Additional research appears to be warranted given the limited amount of information examining waste management derived from shale gas exploration and exploitation.

Land use may also be a highly contested issue among stakeholders in highly populated areas and is often highlighted as a limiting factor for expansion to Europe. As such, the Joint Research Centre (Kavalov and Pelletier 2012) compared the population density in the Barnett play (38 inhabitants per km^2) with the population density of Europe (113 inhabitants per km^2) and concluded that this aspect may be a major barrier for large-scale development of shale gas in the EU.

However, the European Academies Science Advisory Council (EASAC 2014) highlighted that the latest multi-well pads and horizontal drilling techniques reduced building surface areas. These new methods are now commonplace in the industry, even in heavily populated areas such as Pennsylvania, which has a population density similar to most of Europe. Additional research in land use impacts is needed in areas where shale gas is being explored, especially in highly populated areas where conflicting interests between constituents need to be addressed.

Induced seismicity

Induced seismicity refers to earthquakes stimulated by activities where human-introduced stresses are similar in amplitude to the ambient stress state (Rubinstein and Mahani 2015). The link between induced seismicity and human activities (although of small magnitude) have been previously established for reservoir impoundment, conventional oil and gas field depletion, water injection for geothermal energy recovery, and wastewater injections (Davies et al. 2013).

Based on the analyzed articles, it can be seen that studies on induced seismicity were rare between 2010 and 2015, when only eight papers reported on relation to shale gas exploration and exploitation. However, unlike other impacts, three out of eight of these studies were conducted in Europe, a disproportionally large percentage compared to existing exploration there. This may be an indication that regulatory bodies and researchers in Europe are more sensitive to this issue based on a variety of factors, including occurrences of this issue in the USA.

The two main sources of induced seismicity in shale gas exploration and exploitation are hydraulic fracturing and the deep well injection of produced water. As previously mentioned, the link to induced seismicity and deep well injection was previously known, since this is practiced in conventional on shore oil and gas extraction (Rubinstein and Mahani 2015). In the case of hydraulic fracturing, however, researchers initially thought that the volume of fluid used for fracturing, which is significantly lower than the volume disposed of in deep well injection, were unlikely to generate felt seismicity (Clarke et al. 2014).

The larger volume applied in deep well injection in conventional oil and gas is more likely to induce more frequent and larger earthquakes than hydraulic fracturing (McGarr 2014; Rubinstein and Mahani 2015). This counterintuitive observation is mainly due to the fact that both injection volumes and times are significantly lower with hydraulic fracturing when compared to deep well injection, despite higher pressure (McGarr 2014; Rubinstein and Mahani 2015).

Even though researchers originally thought that hydraulic fracturing would not induce felt seismicity for the reasons listed above, this does not apply to every single scenario or study as some report a direct link between the two (Clarke et al. 2014; Holland 2013). For example, low-intensity earthquakes were detected in the UK due to hydraulic fracturing (Clarke et al. 2014; Johnson and Boersma 2013; Stamford and Azapagic 2014) in 2011. This incident marked the first induced seismicity event in Europe associated with shale gas exploration and exploitation and lead to a government suspension of shale gas extraction for 18 months (Clarke et al. 2014; Johnson and Boersma 2013; Stamford and Azapagic 2014).

As a result, the UK now requires the identification of preexisting faults prior to exploration as well as detailed monitoring of induced seismicity during exploration (Milieu 2013). Furthermore, more stringent regulations concerning the threshold for the suspension of operations when compared to other industries were applied to the shale gas industry in the UK (Westaway and Younger 2014). This suggests that there are potentially higher regulatory barriers to shale gas exploration and exploitation in Europe compared to other geographic locations.

Despite these existing studies, there are still many questions and uncertainty between hydraulic fracturing and induced seismicity. One aspect is to examine whether the recent shale gas expansion has led to increased risks of induced seismicity due to the sheer increase in cumulative wastewater volume injected into existing or potentially new disposal wells. However, Rubinstein and Mahani (2015) indicated that the location of the largest increase in seismicity in Oklahoma was not correlated with the deep well injection of spent hydraulic fluids. However, additional studies should examine whether this applies to other plays as well.

An additional aspect concerns the link between induced seismicity by hydraulic fracturing and preexisting faults that was recently established in several articles (Clarke et al. 2014; Frohlich et al. 2011; Holland 2013). This suggests that additional studies that include fault mapping are a potential option to mitigate this issue (Clarke et al. 2014). However, these unmapped faults are often only reactivated after the event occurs, making it difficult to obtain results in advance (Rubinstein and Mahani 2015). In addition, detection methods remain in debate, which may potentially lead to the misidentification or mislabeling of regional natural earthquakes as induced seismicity due to hydraulic fracturing (Caffagni et al. 2014).

Multiple environmental impact assessment

This category encompasses articles that evaluated impacts that could not be placed into a single category (such as health risk assessment from multiple pathways or LCA that incorporate several impacts) or any impact category (for example, socio-economic aspects). The total number of articles in this section was 69 during the analyzed period, with annual percentage ranging from 5 to 35%, and the majority coming from the USA. Further examination showed that none of these was a primary research article (type 1), which indicated that no new data was obtained. Rather, these articles focused on analyzing existing trends.

Concerning multiple impact factor evaluation, LCA is almost always used as the preferred method. This approach was used in several case studies in the UK (Stamford and Azapagic 2014), China (Chang et al. 2015), and the USA (Laurenzi and Jersey 2013). Previous studies using LCA that were referenced in other sections of this review only examined singular compartments rather than a more holistic approach that encompassed multiple environmental aspects for all of the different stages of the life cycle.

Under different scenarios, Stamford and Azapagic (2014) concluded that shale gas may have negative environmental impacts several times higher than conventional NG. This was particularly true for human, marine, freshwater, and terrestrial ecotoxicity. This is one of the few or potentially even the only study that considered impact categories such as acidification potential and element depletion. As a result, this article has been prominently featured in traditional media and in the academic literature, even though it was only published in December 2014. However, it should be noted that the validity of the assumptions and by extension the conclusions in that study remain hotly contested (Stamford and Azapagic 2015; Westaway et al. 2015).

The importance of the LCA approach for a more accurate assessment of the different stages in shale gas exploration and exploitation cannot be overstated. Exploratory LCA may be seen as a tool for decision makers to identify bottlenecks in the process itself and to verify if shale gas production presents more environmental benefits in comparison to other energy sources in a given location. It should be noted that there is a critical lack of specific data, particularly for regions that have yet to be explored, and efforts to close these gaps are needed.

Concluding remarks

There has been an expectable and significant increase in the number of publications on shale gas exploration and exploitation and associated environmental impacts over the years. This is a clear reflection of shale gas production growth in the USA and the increased interest in mirroring this development in other regions, coupled with increase awareness of potential environmental impacts. Although authors from the USA represent a vast majority of the articles examined, several studies from countries which have yet to commercially produce shale gas were found, which suggests a precautionary approach to new regional development.

Regarding existing consensus (Table 2) that seem to emerge from the analysis made in this study, it is important to point out that these may not resist the test of time and are provisional at best. Yet, it is important to identify existing trends in the literature to enable more informed decision and policy makers.

No consensus can be tentatively allocated to air quality; resulting public health risks and land use as results are often

Table 2 List of consensus that emerged from the analysis of this study and relative degree of consensus

Consensus	Relative degree of consensus
- Wastewater characteristics is almost exclusively dependent on rock formation	High
- Migration of methane and salts to groundwater as a result of the fractures rarely occurs	High
- Contamination of surface water as a result of poor wastewater treatment is common	High
- Wastewater organic contaminants tend to be highly biodegradable	Medium
- Wastewater reuse after pretreatment is a simple method to limit negative impacts	High
- Methane leakage percent lies within a 0.66 to 3.9% range	Medium
- Shale gas entire life cycle GHG emissions are lower than coal for electricity generation	High
- Shale gas entire life cycle GHG emissions are lower than coal for heating	Medium
- Seismicity from deep well injection is far more likely than from hydraulic fracturing	High
- Induced seismicity is connected to preexisting faults	Medium

contradictory with no obvious trend, partially because limited studies exist due to the inherent difficulties associated with this type of studies.

As a result, it can be said that more studies within these areas are necessary. However, the observed larger number of studies on water resources might reflect a preliminary identification of this aspect as one of the most sensitive to negative impacts by shale gas exploration and exploitation (and also a bigger public concern). So, more studies on water resources cannot be neglected either.

Aside from the consensus detailed in Table 2, significant reductions in water contamination and treatment requirements and GHG emissions (particularly in well completion) are expected due to new legislation and best industry practices as a result of advances in scientific knowledge and practical experience. Nevertheless, cost-effective wastewater treatment remains a difficult challenge and there are no indications of a solution in the near future, particularly for salt removal. In addition, GHG emission estimations are highly debated as REC technologies have yet to be adequately integrated and characterized.

Finally, LCA appears to be a promising method for a precise overall impact assessment of shale gas though it is currently limited in scope. This may be a reflection on the lack of sufficient raw data due to several propriety aspects and trade secrets of the applied technologies.

Future research efforts should focus on mitigation techniques as well as standardization practices to enable a more precise comparison between studies in order to establish a wider, stronger consensus on environmental impacts of shale gas exploration and exploitation.

Acknowledgements Authors would like to acknowledge IBRASIL (Inclusive and Innovative Brazil project, an Erasmus Mundus Action 2 programme funded by European Commission) for the PhD grant.

References

Abdalla CW, Drohan JR (2010) Marcellus education fact sheet: water withdrawals for development of Marcellus shale gas in Pennsylvania. Penn State Extension, USA

Adair SK, Pearson BR, Monast J, Vengosh A, Jackson RB (2012) Considering shale gas extraction in North Carolina: lessons from other states. Duke Environmental Law and Policy Forum 22:257–301

Ahmadi M, John K (2015) Statistical evaluation of the impact of shale gas activities on ozone pollution in North Texas. Sci Total Environ 536: 457–467. doi:10.1016/j.scitotenv.2015.06.114

Akob DM, Cozzarelli IM, Dunlap DS, Rowan EL, Lorah MM (2015) Organic and inorganic composition and microbiology of produced waters from Pennsylvania shale gas wells. Appl Geochem 60:116–125. doi:10.1016/j.apgeochem.2015.04.011

Akyon B, Stachler E, Wei N, Bibby K (2015) Microbial mats as a biological treatment approach for saline wastewaters: the case of produced water from hydraulic fracturing. Environ Sci Technol 49: 6172–6180. doi:10.1021/es505142t

Alawattegama SK, Kondratyuk T, Krynock R, Bricker M, Rutter JK, Bain DJ, Stolz JF (2015) Well water contamination in a rural community in southwestern Pennsylvania near unconventional shale gas extraction. J Environ Sci Health A Tox Hazard Subst Environ Eng 50:516–528. doi:10.1080/10934529.2015.992684

Allen DT et al (2013) Measurements of methane emissions at natural gas production sites in the United States. Proc Natl Acad Sci U S A 110: 17768–17773. doi:10.1073/pnas.1304880110

Alley B, Beebe A, Rodgers J, Castle JW (2011) Chemical and physical characterization of produced waters from conventional and unconventional fossil fuel resources. Chemosphere 85:74–82. doi:10.1016/j.chemosphere.2011.05.043

Bai B, Carlson K, Prior A, Douglas C (2015) Sources of variability in flowback and produced water volumes from shale oil and gas wells. J Unconv Oil Gas Resour 12:1–5. doi:10.1016/j.juogr.2015.07.001

Baranzelli C, Vandecasteele I, Ribeiro Barranco R, Mari i Rivero I, Pelletier N, Batelaan O, Lavalle C (2015) Scenarios for shale gas development and their related land use impacts in the Baltic Basin. Northern Poland Energy Policy 84:80–95. doi:10.1016/j.enpol.2015.04.032

Barbot E, Vidic NS, Gregory KB, Vidic RD (2013) Spatial and temporal correlation of water quality parameters of produced waters from Devonian-age shale following hydraulic fracturing. Environ Sci Technol 47:2562–2569. doi:10.1021/es304638h

Bogacki M, MacUda J (2014) The influence of shale rock fracturing equipment operation on atmospheric air quality. Arch Min Sci 59: 897–912. doi:10.2478/amsc-2014-0062

Bowen ZH et al (2015) Assessment of surface water chloride and conductivity trends in areas of unconventional oil and gas development—why existing national data sets cannot tell us what we would like to know. Water Resour Res 51:704–715. doi:10.1002/2014WR016382

Brantley SL et al (2014) Water resource impacts during unconventional shale gas development: the Pennsylvania experience. Int J Coal Geol 126:140–156. doi:10.1016/j.coal.2013.12.017

Bunch AG et al (2014) Evaluation of impact of shale gas operations in the Barnett Shale region on volatile organic compounds in air and potential human health risks. Sci Total Environ 468–469:832–842. doi:10.1016/j.scitotenv.2013.08.080

Burnham A, Han J, Clark CE, Wang M, Dunn JB, Palou-Rivera I (2012) Life-cycle greenhouse gas emissions of shale gas. Natural Gas, Coal, Petroleum Environ Sci Technol 46:619–627. doi:10.1021/es201942m

Caffagni E, Eaton D, Van der Baan M, Jones JP (2014) Regional seismicity: a potential pitfall for identification of long-period long-duration events. Geophysics 80:A1–A5. doi:10.1190/GEO2014-0382.1

Cathles Iii LM, Brown L, Taam M, Hunter A (2012) A commentary on “The greenhouse-gas footprint of natural gas in shale formations” by R.W. Howarth, R. Santoro, and Anthony Ingraffea. Clim Chang 113:525–535. doi:10.1007/s10584-011-0333-0

Centner TJ, Petetin L (2015) Permitting program with best management practices for shale gas wells to safeguard public health. J Environ Manag 163:174–183. doi:10.1016/j.jenvman.2015.08.019

Chang Y, Huang R, Masanet E (2014) The energy, water, and air pollution implications of tapping China’s shale gas reserves. Resour Conserv Recycl 91:100–108. doi:10.1016/j.resconrec.2014.07.015

Chang Y, Huang R, Ries RJ, Masanet E (2015) Life-cycle comparison of greenhouse gas emissions and water consumption for coal and shale gas fired power generation in China. Energy 86:335–343. doi:10.1016/j.energy.2015.04.034

Clark CEH, Robert M, Harto CB (2013) Life cycle water consumption for shale gas and conventional natural gas. Environ Sci Technol 47: 11829–11836. doi:10.1021/es4013855

Clarke H, Eisner L, Styles P, Turner P (2014) Felt seismicity associated with shale gas hydraulic fracturing: the first documented example in Europe. Geophys Res Lett 41:8308–8314. doi:10.1002/2014GL062047

Colborn T, Schultz K, Herrick L, Kwiatkowski C (2014) An exploratory study of air quality near natural gas operations. Hum Ecol Risk Assess 20:86–105. doi:10.1080/10807039.2012.749447

Davies R, Foulger G, Bindley A, Styles P (2013) Induced seismicity and hydraulic fracturing for the recovery of hydrocarbons. Mar Pet Geol 45:171–185. doi:10.1016/j.marpetgeo.2013.03.016

Drohan PJ, Brittingham M, Bishop J, Yoder K (2012) Early trends in land cover change and forest fragmentation due to shale-gas development in Pennsylvania: a potential outcome for the north-central Appalachians. Environ Manag 49:1061–1075. doi:10.1007/s00267-012-9841-6

EASAC EASAC (2014) Shale gas extraction: issues of particular relevance to the European Union. Germany

Edwards PM et al (2014) High winter ozone pollution from carbonyl photolysis in an oil and gas basin. Nature 514:351–354. doi:10.1038/nature13767

Edwards RWJ, Celia MA, Bandilla KW, Doster F, Kanno CM (2015) A model to estimate carbon dioxide injectivity and storage capacity for geological sequestration in shale gas wells. Environ Sci Technol 49:9222–9229. doi:10.1021/acs.est.5b01982

Elsner M, Hoelzer K (2016) Quantitative survey and structural classification of hydraulic fracturing chemicals reported in unconventional gas production. Environ Sci Technol 50:3290–3314. doi:10.1021/acs.est.5b02818

Ethridge S, Bredfeldt T, Sheedy K, Shirley S, Lopez G, Honeycutt M (2015) The Barnett Shale: from problem formulation to risk management. J Unconv Oil Gas Resour 11:95–110. doi:10.1016/j.juogr.2015.06.001

Evans JS, Kiesecker JM (2014) Shale gas, wind and water: assessing the potential cumulative impacts of energy development on ecosystem services within the Marcellus play. PLoS One 9. doi:10.1371/journal.pone.0089210

Ferrer I, Thurman EM (2015) Analysis of hydraulic fracturing additives by LC/Q-TOF-MS. Anal Bioanal Chem 407:6417–6428. doi:10.1007/s00216-015-8780-5

Frohlich C, Hayward C, Stump B, Potter E (2011) The Dallas-Fort Worth earthquake sequence: October 2008 through May 2009. Bull Seismol Soc Am 101:327–340. doi:10.1785/0120100131

Gallegos TJ, Varela BA, Haines SS, Engle MA (2015) Hydraulic fracturing water use variability in the United States and potential environmental implications. Water Resour Res 51:5839–5845. doi:10.1002/2015WR017278

Garner J, Cairns J, Read D (2015) NORM in the East Midlands' oil and gas producing region of the UK. J Environ Radioact 150:49–56. doi:10.1016/j.jenvrad.2015.07.016

Getzinger GJ et al (2015) Natural gas residual fluids: sources, endpoints, and organic chemical composition after centralized waste treatment in Pennsylvania. Environ Sci Technol 49:8347–8355. doi:10.1021/acs.est.5b00471

Goetz JD et al (2015) Atmospheric emission characterization of Marcellus Shale natural gas development sites. Environ Sci Technol 49:7012–7020. doi:10.1021/acs.est.5b00452

Gómez VR, Fernández LV, Javier F, Naranjo F (2015) Sustainable alternatives for water management related to shale gas activities in the European context. Madrid

Goodwin RW (2014) Environmental perspective update: hydraulic fracturing. Pollut Eng 46:34–38

Graham J et al (2015) Increased traffic accident rates associated with shale gas drilling in Pennsylvania. Accid Anal Prev 74:203–209. doi:10.1016/j.aap.2014.11.003

Gregory KB, Vidic RD, Dzombak DA (2011) Water management challenges associated with the production of shale gas by hydraulic fracturing. Elements 7:181–186. doi:10.2113/gselements.7.3.181

GWPC GWPC (2009) Modern shale gas development in the United States. A Primer U.S. Department of Energy Office of Fossil Energy National Energy Technology Laboratory, USA

Haghshenas A, Nasr-El-Din HA (2014) Effect of dissolved solids on reuse of produced water at high temperature in hydraulic fracturing jobs. J Nat Gas Sci Eng 21:316–325. doi:10.1016/j.jngse.2014.08.013

Haluszczak LO, Rose AW, Kump LR (2013) Geochemical evaluation of flowback brine from Marcellus gas wells in Pennsylvania. USA Appl Geochem 28:55–61. doi:10.1016/j.apgeochem.2012.10.002

He C, Wang X, Liu W, Barbot E, Vidic RD (2014) Microfiltration in recycling of Marcellus Shale flowback water: solids removal and potential fouling of polymeric microfiltration membranes. J Membr Sci 462:88–95. doi:10.1016/j.memsci.2014.03.035

Heath GA, O' Donoughue P, Arent DJ, Bazilian M (2014) Harmonization of initial estimates of shale gas life cycle greenhouse gas emissions for electric power generation. Proc Natl Acad Sci U S A 111:E3167–E3176. doi:10.1073/pnas.1309334111

Heilweil VM, Grieve PL, Hynek SA, Brantley SL, Solomon DK, Risser DW (2015) Stream measurements locate thermogenic methane fluxes in groundwater discharge in an area of shale-gas development. Environ Sci Technol 49:4057–4065. doi:10.1021/es503882b

Hickenbottom KL, Hancock NT, Hutchings NR, Appleton EW, Beaudry EG, Xu P, Cath TY (2013) Forward osmosis treatment of drilling mud and fracturing wastewater from oil and gas operations. Desalination 312:60–66. doi:10.1016/j.desal.2012.05.037

Hildenbrand ZL et al (2015) A comprehensive analysis of groundwater quality in the Barnett Shale region. Environ Sci Technol 49:8254–8262. doi:10.1021/acs.est.5b01526

Holland AA (2013) Earthquakes triggered by hydraulic fracturing in south-central Oklahoma. Bull Seismol Soc Am 103:1784–1792. doi:10.1785/0120120109

Howarth RW (2014) A bridge to nowhere: methane emissions and the greenhouse gas footprint of natural gas. Energy Sci Eng 2:47–60. doi:10.1002/ese3.35

Howarth RW, Santoro R, Ingraffea A (2011) Methane and the greenhouse-gas footprint of natural gas from shale formations. 106

Howarth RW, Santoro R, Ingraffea A (2012) Venting and leaking of methane from shale gas development: response to Cathles et al. Clim Chang 113:537–549. doi:10.1007/s10584-012-0401-0

IPCC IPoCC (2014) Land use, land-use change and forestry. IPCC >http://www.ipcc.ch/ipccreports/sres/land_use/index.php?idp=160

Jackson RB et al (2013) Increased stray gas abundance in a subset of drinking water wells near Marcellus shale gas extraction. Proc Natl Acad Sci U S A 110:11250–11255. doi:10.1073/pnas.1221635110

Jackson RB, Lowry ER, Pickle A, Kang M, DiGiulio D, Zhao K (2015) The depths of hydraulic fracturing and accompanying water use across the United States. Environ Sci Technol 49:8969–8976. doi:10.1021/acs.est.5b01228

Jaramillo P, Griffin WM, Matthews HS (2007) Comparative life-cycle air emissions of coal, domestic natural gas, LNG, and SNG for electricity generation. Environ Sci Technol 41:6290–6296. doi:10.1021/es063031o

Jiang M, Michael Griffin W, Hendrickson C, Jaramillo P, Vanbriesen J, Venkatesh A (2011) Life cycle greenhouse gas emissions of Marcellus shale gas. Environ Res Lett 6. doi:10.1088/1748-9326/6/3/034014

Johnson C, Boersma T (2013) Energy (in) security in Poland the case of shale gas. Energ Policy 53:389–399. doi:10.1016/j.enpol.2012.10.068

Kassotis CD, Tillitt DE, Davis JW, Hormann AM, Nagel SC (2014) Estrogen and androgen receptor activities of hydraulic fracturing chemicals and surface and ground water in a drilling-dense region. Endocrinology 155:897–907. doi:10.1210/en.2013-1697

Kavalov B, Pelletier N (2012) Shale gas for Europe—main environmental and social considerations: a literature review. European Union, Luxembourg

Kekacs D, Drollette BD, Brooker M, Plata DL, Mouser PJ (2015) Aerobic biodegradation of organic compounds in hydraulic fracturing fluids. Biodegradation 26:271–287. doi:10.1007/s10532-015-9733-6

Kiviat E (2013)Risks to biodiversity from hydraulic fracturing for natural gas in the Marcellus and Utica Shales. 1286. doi:10.1111/nyas.12146

Kolesar Kohl CA, Capo RC, Stewart BW, Wall AJ, Schroeder KT, Hammack RW, Guthrie GD (2014) Strontium isotopes test long-term zonal isolation of injected and Marcellus formation water after hydraulic fracturing. Environ Sci Technol 48:9867–9873. doi:10.1021/es501099k

Latta SC, Marshall LC, Frantz MW, Toms JD (2015) Evidence from two shale regions that a riparian songbird accumulates metals associated with hydraulic fracturing. Ecosphere 6 doi:10.1890/ES14-00406.1

Laurenzi IJ, Jersey GR (2013) Life cycle greenhouse gas emissions and freshwater consumption of Marcellus shale gas. Environ Sci Technol 47:4896–4903. doi:10.1021/es305162w

Lee WJ, Sohn SY (2014) Patent analysis to identify shale gas development in China and the United States. Energ Policy 74:111–115. doi:10.1016/j.enpol.2014.08.009

Lee HJ, Kim HH, Lee H, Lee C (2015) Reaction of aqueous iodide at high concentration with O_3 and O_3/H_2O_2 in the presence of natural organic matter: implications for drinking water treatment. Environ Chem Lett 13:453–458. doi:10.1007/s10311-015-0519-1

Lester Y, Ferrer I, Thurman EM, Sitterley KA, Korak JA, Aiken G, Linden KG (2015) Characterization of hydraulic fracturing flowback water in Colorado: implications for water treatment. Sci Total Environ 512-513:637–644. doi:10.1016/j.scitotenv.2015.01.043

Li J, Jovanovic A, Klimek P, Guo X (2015) Bibliometric analysis of fracking scientific literature. Scientometrics 105:1273–1284. doi:10.1007/s11192-015-1739-7

Litovitz A, Curtright A, Abramzon S, Burger N, Samaras C (2013) Estimation of regional air-quality damages from Marcellus shale natural gas extraction in Pennsylvania. Environ Res Lett 8. doi:10.1088/1748-9326/8/1/014017

Lutz BD, Lewis AN, Doyle MW (2013) Generation, transport, and disposal of wastewater associated with Marcellus shale gas development. Water Resour Res 49:647–656. doi:10.1002/wrcr.20096

Maguire-Boyle SJ, Barron AR (2014) Organic compounds in produced waters from shale gas wells. Environ Sci: Process Impacts 16:2237–2248. doi:10.1039/c4em00376d

Maloney KO, Yoxtheimer DA (2012) Production and disposal of waste materials from gas and oil extraction from the Marcellus shale play in Pennsylvania. Environ Pract 14:278–287. doi:10.1017/S146604661200035X

Manda AK, Heath JL, Klein WA, Griffin MT, Montz BE (2014) Evolution of multi-well pad development and influence of well pads on environmental violations and wastewater volumes in the Marcellus shale (USA). J Environ Manag 142:36–45. doi:10.1016/j.jenvman.2014.04.011

Mauter M, Palmer V (2014) Expert elicitation of trends in Marcellus oil and gas wastewater management. J Environ Eng 140:B4014004. doi:10.1061/(ASCE)EE.1943-7870.0000811

McCawley M (2015) Air contaminants associated with potential respiratory effects from unconventional resource development activities seminars in respiratory. Crit Care Med 36:379–387. doi:10.1055/s-0035-1549453

McGarr A (2014) Maximum magnitude earthquakes induced by fluid injection. J Geophys Res Solid Earth 119:1008–1019. doi:10.1002/2013JB010597

Middleton RS et al (2015) Shale gas and non-aqueous fracturing fluids: opportunities and challenges for supercritical CO_2. Appl Energy 147:500–509. doi:10.1016/j.apenergy.2015.03.023

Milieu L (2013) Regulatory provisions governing key aspects of unconventional gas extraction in selected Member States, Brussels

Molofsky LJ, Connor JA, Farhat SK, Wylie AS Jr, Wagner T (2011) Methane in Pennsylvania water wells unrelated to Marcellus shale fracturing. Oil Gas J 109:54–67 +93

Moran M, Cox AB, Wells R, Benichou C, McClung M (2015) Habitat loss and modification due to gas development in the Fayetteville shale. Environ Manag 55:1276–1284. doi:10.1007/s00267-014-0440-6

Mykowska A, Rogala A, Kallas A, Karczewski J, Hupka J (2015) Radioactivity of drilling cuttings from shale resources of the lower Paleozoic Baltic basin. Physicochem Problems Miner Process 51:521–533. doi:10.5277/ppmp150213

Nelson AW, Knight AW, Eitrheim ES, Schultz MK (2015) Monitoring radionuclides in subsurface drinking water sources near unconventional drilling operations: a pilot study. J Environ Radioact 142:24–28. doi:10.1016/j.jenvrad.2015.01.004

Nicot JP, Scanlon BR (2012) Water use for shale-gas production in Texas, U.S. Environ Sci Technol 46:3580–3586. doi:10.1021/es204602t

Nicot JP, Scanlon BR, Reedy RC, Costley RA (2014) Source and fate of hydraulic fracturing water in the Barnett shale: a historical perspective. Environ Sci Technol 48:2464–2471. doi:10.1021/es404050r

Olmstead SM, Muehlenbachs LA, Shih J-S, Chu Z, Krupnick AJ (2013) Shale gas development impacts on surface water quality in Pennsylvania. Proc Natl Acad Sci 110:4962–4967. doi:10.1073/pnas.1213871110

Orem W et al (2014) Organic substances in produced and formation water from unconventional natural gas extraction in coal and shale. Int J Coal Geol 126:20–31. doi:10.1016/j.coal.2014.01.003

Osborn SG, Vengosh A, Warner NR, Jackson RB (2011) Methane contamination of drinking water accompanying gas-well drilling and hydraulic fracturing. Proc Natl Acad Sci U S A 108:8172–8176. doi:10.1073/pnas.1100682108

OSHA OSHA, NIOSH NIoOSaH (2015) Worker exposure to silica during hydraulic fracturing. DOL, Departament of Labor, USA

O, Sullivan F, Paltsev S (2012) Shale gas production: potential versus actual greenhouse gas emissions. Environ Res Lett 7. doi:10.1088/1748-9326/7/4/044030

Pacsi AP, Sanders KT, Webber ME, Allen DT (2014) Spatial and temporal impacts on water consumption in Texas from shale gas development and use. ACS Sustain Chem Eng 2:2028–2035. doi:10.1021/sc500236g

Pancras JP, Norris GA, Landis MS, Kovalcik KD, McGee JK, Kamal AS (2015) Application of ICP-OES for evaluating energy extraction and production wastewater discharge impacts on surface waters in western Pennsylvania. Sci Total Environ 529:21–29. doi:10.1016/j.scitotenv.2015.04.011

Paulik LB et al (2015) Impact of natural gas extraction on PAH levels in ambient air. Environ Sci Technol 49:5203–5210. doi:10.1021/es506095e

Peischl J et al (2015) Quantifying atmospheric methane emissions from the Haynesville, Fayetteville, and northeastern Marcellus shale gas production regions. J Geophys Res D: Atmospheres 120:2119–2139. doi:10.1002/2014JD022697

Prpich G, Coulon F, Anthony EJ (2016) Review of the scientific evidence to support environmental risk assessment of shale gas development in the UK. Sci Total Environ 563-564:731–740. doi:10.1016/j.scitotenv.2015.11.026

Racicot A, Babin-Roussel V, Dauphinais JF, Joly JS, Noël P, Lavoie C (2014) A framework to predict the impacts of shale gas infrastructures on the forest fragmentation of an agroforest region. Environ Manag 53:1023–1033. doi:10.1007/s00267-014-0250-x

Reap E (2015) The risk of hydraulic fracturing on public health in the UK and the UK's fracking legislation. Environ Sci Eur 27. doi:10.1186/s12302-015-0059-0

Richardson N, Gottlieb M, Krupnick A, Wiseman H (2013) The state of state shale gas regulation. RFF, Resources for the Future, Washington, DC

Rodriguez RS, Soeder DJ (2015) Evolving water management practices in shale oil & gas development. J Unconv Oil Gas Resour 10:18–24

Rosenman KD (2014) Hydraulic fracturing and the risk of silicosis. Clin Pulm Med 21:167–172. doi:10.1097/CPM.0000000000000046

Roy AA, Adams PJ, Robinson AL (2014) Air pollutant emissions from the development, production, and processing of Marcellus shale natural gas. J Air Waste Manage Assoc 64:19–37. doi:10.1080/10962247.2013.826151

Rubinstein JL, Mahani AB (2015) Myths and facts on wastewater injection, hydraulic fracturing, enhanced oil recovery, and induced seismicity. Seismol Res Lett 86:1060–1067. doi:10.1785/0220150067

Rutter AP et al (2015) Sources of air pollution in a region of oil and gas exploration downwind of a large city. Atmos Environ 120:89–99. doi:10.1016/j.atmosenv.2015.08.073

Shank MK, Stauffer JR (2015) Land use and surface water withdrawal effects on fish and macroinvertebrate assemblages in the Susquehanna River basin, USA. J Freshw Ecol 30:229–248. doi:10.1080/02705060.2014.959082

Shih JS, Saiers JE, Anisfeld SC, Chu Z, Muehlenbachs LA, Olmstead SM (2015) Characterization and analysis of liquid waste from Marcellus shale gas development. Environ Sci Technol 49:9557–9565. doi:10.1021/acs.est.5b01780

Skalak KJ, Engle MA, Rowan EL, Jolly GD, Conko KM, Benthem AJ, Kraemer TF (2014) Surface disposal of produced waters in western and southwestern Pennsylvania: potential for accumulation of alkali-earth elements in sediments. Int J Coal Geol 126:162–170. doi:10.1016/j.coal.2013.12.001

Song W et al (2015) A multiyear assessment of air quality benefits from China's emerging shale gas revolution: Urumqi as a case study. Environ Sci Technol 49:2066–2072. doi:10.1021/es5050024

Stamford L, Azapagic A (2014) Life cycle environmental impacts of UK shale gas. Appl Energy 134:506–518. doi:10.1016/j.apenergy.2014.08.063

Stamford L, Azapagic A (2015) Response to the comment by Westaway et al. (applied energy 148 (2015) 489–495) on the paper "Life cycle environmental impacts of UK shale gas" by Stamford and Azapagic (applied energy 134 (2014) 506–518). Appl Energy 155:947–948. doi:10.1016/j.apenergy.2015.05.020

Stearman LW, Adams G, Adams R (2014) Ecology of the redfin darter and a potential emerging threat to its habitat. Environ Biol Fish 98:623–635. doi:10.1007/s10641-014-0300-5

Steliga T, Kluk D, Jakubowicz P (2015) Analysis of chemical and toxicological properties of fluids for shale hydraulic fracturing and flowback water. Pol J Environ Stud 24:2185–2196. doi:10.15244/pjoes/43501

Stephenson T, Valle JE, Riera-Palou X (2011) Modeling the relative GHG emissions of conventional and shale gas production. Environ Sci Technol 45:10757–10764. doi:10.1021/es2024115

Swarthout RF et al (2015) Impact of Marcellus shale natural gas development in southwest Pennsylvania on volatile organic compound emissions and regional air quality. Environ Sci Technol 49:3175–3184. doi:10.1021/es504315f

Thorn TH (2015) Shale gas environmental issues and challenges. In: Rezaee R (ed) Fundamentals of shale gas reservoirs. Wiley, USA, pp 381–395

Thurman EM, Ferrer I, Blotevogel J, Borch T (2014) Analysis of hydraulic fracturing flowback and produced waters using accurate mass: Identification of ethoxylated surfactants. Anal Chem 86:9653–9661. doi:10.1021/ac502163k

USEIA USEIA (2013) Technically recoverable shale oil and shale gas resources: an assessment of 137 shale formations in 41 countries outside the United States. USEIA, USA

USEIA USEIA (2015) Natural gas. USEIA http://www.eia.gov/naturalgas/data.cfm

USEPA USEPA (2010) Potential relationships between hydraulic fracturing and drinking water resources EPA, USA

USEPA USEPA (2011) Plan to study the potential impacts of hydraulic fracturing on drinking water resources USEPA, Washington

USEPA USEPA (2015a) The hydraulic fracturing water cycle. USEPA http://www.epa.gov/hfstudy/hydraulic-fracturing-water-cycle. 2015

USEPA USEPA (2015b) Review of state and industry spill data: characterization of hydraulic fracturing-related spills vol EPA/601/R-14/001.

USEPA USEPA (2016) Regulatory actions. EPA https://www3.epa.gov/airquality/oilandgas/actions.html

USGPO USGPO (2016) Effluent limitations guidelines and standards for the oil and gas extraction point source category.

Walter GR, Benke RR, Pickett DA (2012) Effect of biogas generation on radon emissions from landfills receiving radium-bearing waste from shale gas development. J Air Waste Manage Assoc 62:1040–1049. doi:10.1080/10962247.2012.696084

Wang Q, Li R (2016) Natural gas from shale formation: a research profile. Renew Sust Energ Rev 57:1–6. doi:10.1016/j.rser.2015.12.093

Wang J, Ryan D, Anthony EJ (2011) Reducing the greenhouse gas footprint of shale gas. Energ Policy 39:8196–8199. doi:10.1016/j.enpol.2011.10.013

Wang H, Li G, Shen Z (2012) A feasibility analysis on shale gas exploitation with supercritical carbon dioxide. Energy Sources, Part A: Recovery, Utilization, and Environmental Effects 34:1426–1435. doi:10.1080/15567036.2010.529570

Warner NR et al (2012) Geochemical evidence for possible natural migration of Marcellus formation brine to shallow aquifers in Pennsylvania. Proc Natl Acad Sci U S A 109:11961–11966. doi:10.1073/pnas.1121181109

Warner NR, Christie CA, Jackson RB, Vengosh A (2013a) Impacts of shale gas wastewater disposal on water quality in western Pennsylvania. Environ Sci Technol 47:11849–11857. doi:10.1021/es402165b

Warner NR, Kresse TM, Hays PD, Down A, Karr JD, Jackson RB, Vengosh A (2013b) Geochemical and isotopic variations in shallow groundwater in areas of the Fayetteville shale development, north-central Arkansas. Appl Geochem 35:207–220. doi:10.1016/j.apgeochem.2013.04.013

Westaway R, Younger PL (2014) Quantification of potential macroseismic effects of the induced seismicity that might result from hydraulic fracturing for shale gas exploitation in the UK. Q J Eng Geol Hydrogeol 47:333–350

Westaway R, Younger PL, Cornelius C (2015) Comment on "Life cycle environmental impacts of UK shale gas" by L. Stamford and A. Azapagic. Appl Energy, 134, 506–518, 2014 Applied Energy 148:489-495 doi:10.1016/j.apenergy.2015.03.008

Ziemkiewicz PF, Thomas He Y (2015) Evolution of water chemistry during Marcellus shale gas development: a case study in West Virginia. Chemosphere 134:224–231. doi:10.1016/j.chemosphere.2015.04.040