

Research article

Estimation of the water cycle related to shale gas production under high data uncertainties: Dutch perspective

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ABSTRACT

The potential water demand for fracturing fluids along with the possible flowback and produced water production is assessed for the Dutch Posidonia shale. Total water demand estimated for 25 years of the field development using historic data from the U.S. plays varies between 12.2 and 36.9 Mm³. The maximal annual water consumption of 0.95–2.88 Mm³ is expected in the peak years of shale gas production. These figures are much lower than the availability of any potential water sources, which include drinking water, fresh and brackish groundwater, river water, effluents of wastewater treatment plants (WWTP) and sea water. River water is considered the most promising water source for fracturing fluids in the Dutch Posidonia shale based on its availability ($> 6 \cdot 10^4$ Mm³/year) and quality (only bacterial composition needs to be controlled). Total wastewater production for the whole period of the field development is estimated between 6.6 and 48.0 Mm³. Wastewater recycling can cover significant part of the source water demand for fracturing fluid. However, high mineral content of the wastewater as well as temporal and spatial discrepancies between wastewater production and water demand will form obstacles for wastewater recycling. The assessment framework developed in this study may be applied for other shale gas fields with high uncertainties regarding subsurface properties, connate formation water characteristics and future legislative framework.

1. Introduction

Potential upcoming production of natural gas from unconventional resources has become a highly debated topic in Europe in recent years. The debate has been sparked by the example set by the United States, where shale gas accounted for 47% of total dry gas production in 2015 (EIA, 2015b). Economic merits, security of provision, solution of geopolitical tensions, and a possible reduction of greenhouse gas emissions are relevant motives for the exploration and production of unconventional gas (Howarth et al., 2011; Jenner and Lamadrid, 2013; Laurenzi and Jersey, 2013). However, numerous environmental concerns, including negative impacts on water resources, air and soil quality as well as probability for man-induced earthquakes challenge the expediency of shale gas production (Hays et al., 2015; Howarth et al., 2011; Kargbo et al., 2010; Small et al., 2014; Soeder et al., 2014).

Although not limited to shale gas production solely, hydraulic fracturing is the most controversial aspect in the debates around shale gas. In combination with horizontal drilling this technology is used to

mine gas from impermeable shale layers with limited gas mobility. The well is drilled vertically till the depth of the shale layer is reached, afterwards horizontal drilling is performed (Wang et al., 2014). Horizontal drilling allows for an increased contact area with shale and decreases the number of vertical wells required for development of a single play. Thousands of cubic meters of water mixed with chemicals and proppant are pumped into the horizontal wells under pressure in order to create micropores in the shale, thus increasing permeability of the formation and mobility of the captured gas (Gregory et al., 2011). The process known as hydraulic fracturing causes numerous concerns related to particularly large quantities of water and chemicals used for the process, possible groundwater and surface water contamination and challenges related to wastewater treatment (Arthur and Coughlin, 2011; Brantley et al., 2014; Small et al., 2014; Vidic et al., 2013). Shale gas production requires significant amounts of water for fracturing fluid formulations, with values between 1000 m³/well and 38 000 m³/well reported in the literature (Chen and Carter, 2016; Eaton, 2013; Kargbo et al., 2010). Although shale gas is not a water-intensive fuel compared

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to other energy sources, local impact on water resources may vary depending on the availability of water sources and competing water withdrawals (Goodwin et al., 2014; Nicot and Scanlon, 2012). Large quantities of wastewater with high salinity (up to 200 g Cl⁻/L), complex organic composition and potentially elevated radioactivity emissions may cause serious environmental problems (Annevelink et al., 2016; Gregory et al., 2011; Howarth et al., 2011). Wastewater from shale gas wells is often categorized by operators into flowback and produced water. The first term is related to initial wastewater flow after the well commencement, which is characterised by high flow rates and comparatively low salinity, while the second term denotes long-term wastewater flow from the well, characterised by moderate flow rates and high salinity (Gregory et al., 2011; Nicot and Scanlon, 2012; Shih et al., 2015). No strict distinction between both terms exists; therefore general term “wastewater” is used further in the paper to denote both flows.

The main experience in the environmental management related to shale gas production has been gained in the United States so far. The geologic, geographic, climatic and economic conditions, as well as the legislative framework of the United States differ from those in Europe (Faber et al., 2017). Prediction of the environmental impact of shale gas production in Europe has to be extrapolated from the experience gained from the cases in the U.S. and the scarce data on the properties of potential shale gas fields.

This paper aims at the initial assessment of the water management strategies required for the Posidonia shale gas field, which has the highest development potential in the Netherlands. Despite the temporal ban on shale gas extraction extended by the government till 2023, possible scenarios for shale gas production have to be developed and evaluated. In this paper the framework for the estimation of the source water requirements and wastewater volumes is established and possible management strategies are proposed. In addition, the coverage of water demand for fracturing fluid by flowback and produced water (FPW) recycling is assessed.

2. Materials and methods

2.1. Study area

The shale gas formations in the Netherlands include the Posidonia formation and the Geverik member of the Epen formation (PlanMER, 2015). The Posidonia formation stretches across the south of the Netherlands from Germany to the North Sea, with core areas located in the province of North Brabant (Fig. 1). Having the core area depth of 3500 m, gross thickness of 15–35 m and an average total organic carbon content of 6% the Posidonia formation has a potential for a future industrial shale gas production (van Bergen et al., 2013).

The Geverik member is generally located at the depth > 4000 m which makes large scale shale gas production questionable (van Bergen et al., 2013). Moreover, information on the geological and petrological properties of the Geverik member of the Epen formation is scarce. Therefore, water-related issues associated with shale gas production from the Geverik member of the Epen formation are not considered in this study.

2.2. Estimation of the area with potential for shale gas production

The area of the shale which can be developed without constraints is smaller than the total area of the shale. The potential for shale gas accumulation depends on the thermal maturity of the shale, hence immature and overmature shales do not have sufficient gas-forming potential and are not considered as areas for prospective shale gas production in this study. Some areas are also excluded due to legislative constraints; the functions of others are not complementary with the shale gas production process. Urban areas, Natura 2000-zones, water production and groundwater protection zones, and large water courses

are to be excluded from prospective shale gas production areas in the Netherlands based on the legislation and land availability (PlanMER, 2015).

The maps of geographic location and maturity of Posidonia shale, Dutch urban areas and Natura 2000-zones were imported to ArcMAP 10.3 software as.shp files from open sources (Kadaster, 2016; Natura-2000, 2013; NLOG, 2014). The map of water production and groundwater protection zones was obtained from the author with permissions (van der Aa et al., 2015).

Negative effects of shale gas production on the air quality and human health are generally observed within a distance of 1 km from the well heads (Hill, 2014; McKenzie et al., 2012). A ban for well drilling within 1 km of urban areas and Natura 2000-zones is therefore a likely regulatory measure to be imposed (PlanMER, 2015). Hence a positive buffer of 1000 m was added to the .shp maps of Dutch urban areas and Natura-2000 zones. On the other hand, horizontal drilling allows development of the reserves under areas with drilling constrains from remote vertical wells. Horizontal laterals with length up to 3000 m can be commenced with currently available technologies (Nicot et al., 2014), however, drilling of that long laterals is not always possible. Hence commencement of wells with 2500 m laterals under the areas with drilling constrains is assumed and a negative buffer of 2500 m was added to all .shp maps of the areas with drilling constrains. All areas with immaturity, overmaturity and drilling constrains were subtracted from the total onshore area of the Posidonia shale within the borders of the Netherlands and the resulting area was calculated in ArcMAP 10.3 (Fig. 1).

The impact of horizontal drilling with fracturing on drinking water sources is a highly debated topic, with no common scientific opinion on the question (EPA, 2016a; MIM and MEZ, 2016; Olmstead et al., 2013; Osborn et al., 2011; Vidic et al., 2013; Warner et al., 2012). Legislative ban on the horizontal drilling under water production and groundwater protection zones is likely to be imposed (PlanMER, 2015). To account for this ban, water production and groundwater protection zones were subtracted from the previously estimated area with potential for shale gas production.

2.3. Estimation of the water requirements for hydraulic fracturing

The volume of water required for fracturing of a single well depends on a number of factors; i.e. geology, formation depth and lateral length among them (Rahm and Riha, 2014). However respective data for the Posidonia formation are scarce due to limited exploration of the play. Data obtained during exploration and exploitation of shale plays in the United States will therefore be extrapolated to the Posidonia formation taken into account available data on the formation properties. Two scenarios for estimation of water consumption for hydraulic fracturing are applied and compared in this study, namely estimations based on the historical water consumption (i) per single well and (ii) per unit of lateral length of the horizontal drills. It is not possible to relate the estimated water volumes to the amount of gas produced, because the estimations of the recoverable gas reserves for the undeveloped formations often have at least one order of magnitude difference.

Scenario 1 is based on the estimation of the average water volume required for fracturing of a single well using historical data and extrapolating the water consumption on the whole play with account for an expected well intensity. The U.S. Geological Survey National Produced Waters Geochemical Database which contains data from ca. 1.8 million hydraulic fracturing treatments and ca. 1 million wells in the United States was used for the estimation (Gallegos et al., 2015; USGS, 2014). Only the entries for wells with shale gas production applying horizontal drilling and hydraulic fracturing in 2000–2010 were considered (15,742 wells in total), because horizontal drilling was rarely used prior to 2000.

The Posidonia formation is rich in clay and relatively ductile (van Bergen et al., 2013). Gel fluids are usually applied for fracturing of the

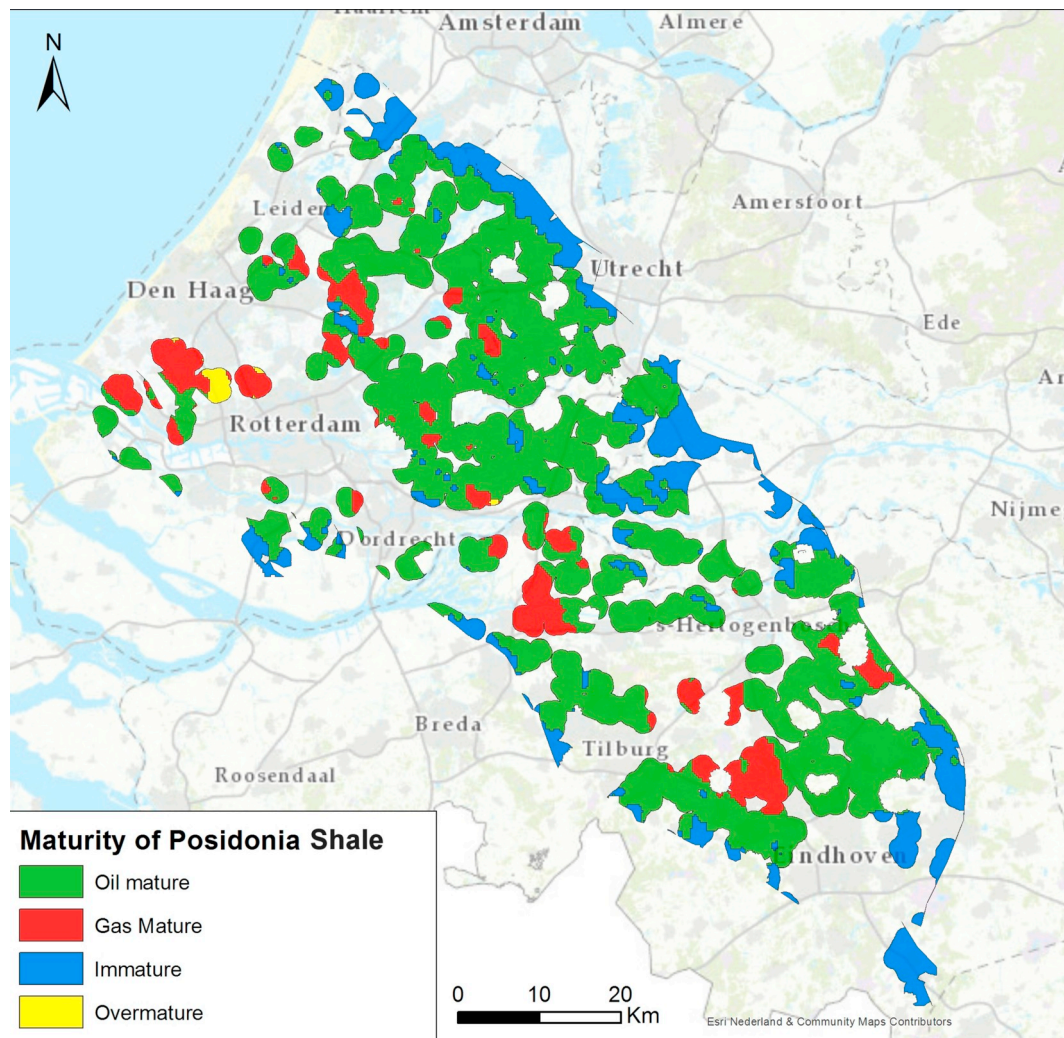


Fig. 1. Location of the Posidonia formation within the borders of the Netherlands and the maturity of the areas with potential for shale gas production.

formations with such properties (Barati and Liang, 2014). The water consumption for single well given in the USGS database is not paired to the type of fluid used. However, the number of fracturing activities performed with specific fluids is also given in the aforementioned database. According to these data the use of slick water as fracturing fluid was insignificant prior to 2007 (USGS, 2014). Hence it is assumed that the average water volume required for fracturing of a single well in 2000–2006 is representative for gel-fluid fractures.

According to the study on the estimation of the potential for shale gas production in North Brabant, 329 wells are expected to be commenced within 330.5 km², resulting in the well density of 1 well/km² (Halliburton, 2011). The same well density is assumed in this study for the areas with high potential for gas production, or gas mature areas (W_g), whereas the well density 0.5 well/km² is assumed for the areas with high potential for oil production, or oil mature areas (W_o) (Cirkel et al., 2014). The latter areas also have potential for gas production, though lower than the gas mature areas. The total volume of water required for the development of formation (Sw_{tot}) is calculated from an average volume of water consumed by a single horizontal well in 2000–2006 (V , m³/well), gas and oil mature areas of formation (A_g and A_o , km²), and respective well densities:

$$Sw_{tot} = V(A_g W_g + A_o W_o) \tag{1}$$

Scenario 2 is based on the calculation of total lateral lengths within the formation and application of an average water use intensity (WUI) factor, which is a median water use per unit of lateral length (Nicot and

Scanlon, 2012) (2.2). The total lateral length is estimated from the average lateral spacing (400 m) applied in the shale gas production areas with horizontal drilling (Halliburton, 2011), which results in the lateral density of 2.5 km/km² in the gas mature areas (Dl_g). Twice lower lateral density is assumed for oil mature areas ($Dl_o = 1.25$ km/km²). Based on the relatively narrow range (9.5–14 m³/m) for the median WUI for the three U.S. shale plays with different geological properties (Barnett, Haynesville and Eagle Ford) Nicot and Scanlon (2012) proposed to consider the median value for these three shale plays for the estimation of water consumption of newly developed formations. Therefore the WUI of 12 m³/m is considered for estimation of water consumption in this study.

$$Sw_{tot} = WUI(A_g Dl_g + A_o Dl_o) \tag{2}$$

Both scenarios exclude water consumption for well drilling, which, according to the literature, requires 300–380 m³ or less than 3% of the median water use for fracturing (Chen and Carter, 2016; Jiang et al., 2014). The indirect water consumption for production of proppant and chemical additives is also excluded from the calculations, because of significant uncertainties regarding the composition of fracturing fluids and allocation of the production facilities.

Old wells might be refractured to increase gas mobility, which could potentially influence the temporal trends of water consumption within the shale play. However, refracturing is seldom applied so far for shale, corresponding to 0.22% of all fracturing operations listed in USGS database. Consequently it is also excluded from the estimation (USGS,

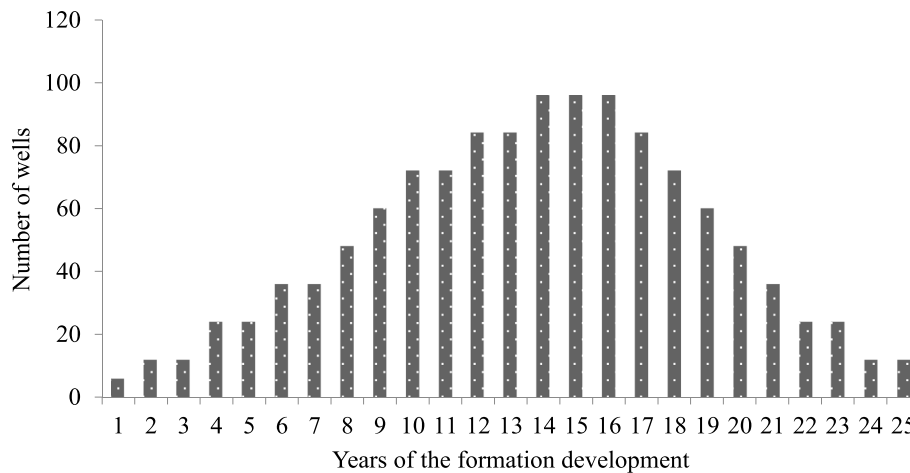


Fig. 2. Assumed temporal development of the Posidonia shale within the Dutch borders.

2014).

The annual water requirements for shale gas production in the on-shore Dutch Posidonia shale (Sw_p) are estimated assuming 25 years of the play development (Fig. 2). The lag-phase of gas production is expected at the beginning of shale play development based on the technological and operational constraints (Gény, 2010; Kavalov and Pelletier, 2012; Rogers, 2011). It is therefore assumed that the shale gas exploitation starts relatively slowly (Tian et al., 2014), i.e. with 6 wells/year (or 0.5 wells/month) being put into operation during the first year and 12 wells/year (or 1 well/month) during the next two years. An increase in the number of new wells cumulates in the years 14–16, when 96 new wells are put into operation annually or 8 new wells monthly. Then a steep decrease in the drilling activities from year 17 till year 25 is assumed (Fig. 2).

2.4. Estimation of the available water reserves

The available drinking water reserves are defined as the difference between necessary and normative water production capacities of the five drinking water treatment companies, which operate in the area of Posidonia shale (Brabant Water, Dunea, Evides (South Holland), Oasen and Vitens (Utrecht)) (Tangena, 2014). Necessary water production capacity accounts for the net drinking water demand, and include production and distribution losses and margins for unexpected water demand increase. Normative water production capacity is determined by the permit, winning and purification capacities of the production company.

The groundwater reserves are estimated as the difference between the existing withdrawal permits and the necessary water production capacities of the abovementioned drinking water production companies as per 2015 (van der Aa et al., 2015). Fresh water reserves are estimated from the annual discharge of the three biggest rivers in the region, the Waal, the Lek and the Meuse, measured at Tiel, Hagestein and Keizersveer, respectively in 2016 (Rijkswaterstaat, 2017). Smaller rivers are excluded from estimation of surface water reserves due to the high anthropogenic pressure which is already put on these water sources and complicated regulations regarding potential water extraction. Volumes of wastewater treatment plant (WWTP) effluents are estimated from the annual municipal wastewater production in the Maas watershed area in 2016 (CBS, 2017).

2.5. Estimation of wastewater volumes

Several authors estimated wastewater volumes from shale gas wells as the percentage of injected fracturing fluid. According to Kondash et al. (2017), who analysed long-term data on water consumption and

production at Barnett (10 years period), Eagle Ford and Haynesville (7 years period) formations, median wastewater volumes correspond to 45–62% of the injected fracturing fluid. In contrast, Nicot et al. (2014) estimated the median wastewater volumes at Barnett formation exceeding volumes of injected fluid after several years of operation and reaching 110% of injected fluid volume. The deviations between the studies may be caused by the different data sources used. To cover the range of variations caused by these possible deviations total wastewater volumes from single well at Posidonia formation assumed for the further calculations in this study were 45% (low recovery scenario) and 110% (high recovery scenario) of the injected fracturing fluid respectively. The well lifespan of 10 years was assumed according to the current trends observed in the U.S. (EPA, 2016a; Nicot et al., 2014).

Kondash et al. (2017) also presented the median wastewater production at Barnett, Haynesville and Eagle Ford formations during first 1, 3 and 6 months as the percentage of total wastewater production. These data of Kondash et al. (2017) were fitted to logarithmic function ($V(t) = a \ln(t)$), where $V(t)$ is the percentage of recovered fracturing fluid at time t , assuming low (45%) and high (110%) total wastewater recovery (Fig. S1). The standard deviation of the coefficient a was below 3%, therefore logarithmic functions with the average values of a (9.55 and 23.35 for low and high wastewater recovery scenarios respectively) were used to estimate the percentage of total wastewater recovery for every month of the well lifespan. The percentage of monthly recovered wastewater was defined as the difference between the percentages of total recovered wastewater at time t , month and time $(t - 1)$, month. The data obtained were fitted to the power functions, which were used to estimate the percent of monthly recovered wastewater for every month of the well lifespan (Fig. S2). New wells are introduced with a time delay dt_i , which can be inferred from Table S1. Specifically, wastewater production profile for i^{th} well can be described by eq. (3):

$$Pw_i(t) = Sw_{tot} C \cdot (t - dt_i)^{-\alpha}, 0 + dt_i < t \leq 120 + dt_i \quad (3)$$

where C and α are empirical coefficients characterising wastewater production by the well ($C = 0.1045$, $\alpha = 1.018$; $C = 0.2493$, $\alpha = 1.013$ for wastewater production at low and high wastewater recovery respectively (Fig. S2). Wastewater produced by all N wells at any time t (Pw_t) is given by eq. (4):

$$Pw(t) = \sum_{i=1}^N Pw_i(t) \quad (4)$$

Wastewater produced by all N wells in a time period between t_0 and t_f , (Pw_{tot}) as well as the annual wastewater production (Pw_y) is given by eq. (5):

Table 1Comparison of predicted water consumption and available water reserves from different sources within the Dutch Posidonia shale (in Mm³/y).

Maximal predicted annual water consumption		Available water reserves				
Low estimate	High estimate	Drinking water - current supply	Drinking water - predicted supply	Ground water	River water	Wastewater (The Meuse watershed district)
0.95	2.88	45.4 ^a	27.3 ^a	75.5 ^b	63835 ^c	431 ^d

^a Based on the data of necessary and normative production capacities by Tangena (2014).^b Estimated from van der Aa et al. (2015).^c Sum of the average water discharge of The Lek at Hagestein, The Waal at Tiel and The Meuse at Keizersveer.^d CBS (2017).

$$Pw_{tot} = \sum_{i=1}^N \sum_{t=t_0}^{t_f} Pw_i(t) \quad (5)$$

Both values of Sw_{tot} estimated from the historical water consumption for single well (Scenario 1) and for the unit of lateral length (Scenario 2) were used for estimation of wastewater volumes according to Eq. (3). Together with two different wastewater recovery scenarios these resulted in four different scenarios of estimated wastewater production.

2.6. Wastewater recycling and source water consumption

Wastewater recycling reduces water consumption from other sources. This reduction can be calculated according to eq. (6):

$$\begin{cases} Sr_y = Sw_y - Pw_y \cdot r, & \text{if } Pw_y \cdot r \leq Sw_y \\ Sr_y = 0, & \text{if } Pw_y \cdot r > Sw_y \end{cases} \quad (6)$$

where Sr_y is annual reduced source water consumption, Pw_y – annual wastewater production, Sw_y – annual source water consumption, r – recycling ratio. Flowback and produced water recycling in the U.S. varies greatly depending on the availability of other options for wastewater management and economic feasibility ranging from 25% recycling at Eagle Ford to 90% recycling at Marcellus in 2012–2014. Thus 25% and 90% recycling of the annually produced wastewater are considered as low and high recycling scenarios respectively.

3. Results and discussion

3.1. Required water volumes and potential water sources

3.1.1. Estimated water consumption

The total area of the onshore Posidonia shale within the Dutch borders estimated using the ArcGIS software is 5198 km². The area, which can be developed without constraints from surface land use is 2497 km² when horizontal drilling under water production and groundwater protection areas is banned, and it is 2668 km² when development of these areas is allowed. The difference between the two production areas calculated under this different legislative scenarios is < 7%. Therefore prohibition of shale development under water production and groundwater protection zones within onshore Dutch Posidonia shale will only have a minor impact on the volumes of fracturing fluid, flowback and produced water at a country level. Therefore only the scenario with horizontal drilling allowed under the drinking water protection zones is further considered in the study.

Only 293 km² of the area without surface constraints is gas mature. The area with sufficient maturity for oil production is much higher (1872 km²). If the well intensity of 1 well/km² for gas mature and 0.5 well/km² for oil mature areas is considered, 1230 wells will be drilled during the whole period of the shale development.

Estimated total water consumption over 25 years of the shale development calculated as a function of the average historical water consumption per single well is 12.2 Mm³ (Scenario 1); as a function of

the average lateral length – 36.9 Mm³ (Scenario 2). Scenario 1 is expected to give the underestimated value, because it is based on the historical data sets of 2000–2006, whereas general trends show increasing annual water use per well related to the increasing length of laterals and numbers of fracturing stages (Chen and Carter, 2016). Scenario 2 gives overestimated value, because complete development of the whole prospective area with the lateral density of 2.5 km/km² for gas mature zones and 1.25 km/km² for oil mature zones is highly unlikely. It will be hindered by the constraints, which are not included in the current scenarios, e.g., those related to the land ownership (Baranzelli et al., 2015). Therefore, water consumption of 12.2 Mm³ calculated according to the Scenario 1 and 36.9 Mm³ calculated according to the Scenario 2 are considered as indicative values for minimal and maximal estimated water consumption (Table S2), and it can be concluded that uncertainty bands are only a factor of 3.

The estimated cumulative annual water consumption under assumption that all wells consume equal amount of water and have an equal lateral length is 0.95 Mm³ and 2.88 Mm³ for Scenario 1 and Scenario 2 respectively in the years with maximal play development (96 wells/y).

3.1.2. Quantity of potential water sources

The sources which may cover the water demand for hydraulic fracturing include drinking water supply, groundwater, fresh surface water, sea water, and WWTP effluents. The data on estimated cumulative annual water consumption and available water reserves, except for sea water, which is considered an unlimited source, are shown in Table 1. Current data (2015) and estimated data for 2030 are shown for the available drinking water reserves. The drinking water reserves in 2030 are estimated from expected population increase and external factors, such as source pollution, climate change and environmental policy (see Tangena, 2014 for the details).

All considered water sources can singly cover the estimated demand for shale gas production. River water is the most abandoned source, with 0.0014%–0.0045% of annual discharge required to cover water demand for shale gas production in the years with maximal play development. Drinking water is the least abandoned source, with 3.5–10.5% of the available reserves required to cover shale gas production demand.

3.1.3. Quality of potential water sources

Fracturing fluids require a certain quality of source water to ensure the absence of interferences with the fluid additives. Sulphate, calcium, magnesium, barium, strontium, iron, bicarbonate, and high salt concentrations can cause scaling on the equipment surfaces and in the borehole, interfere with the fluid additives and promote growth of microorganisms (Haghshenas and Nasr-El-Din, 2014; Lutz et al., 2013; Sun et al., 2012). Furthermore, high counts of microorganisms in the source water are undesired because they can cause bacterial fouling or corrosion (EPA, 2016b; Lutz et al., 2013).

The source water composition which is suitable for fracturing fluid formulations and does not lead to scaling, fouling, corrosion or

Table 2
Recommended source water quality for hydraulic fracturing operations and quality of the potential water sources.

Parameter	Recommended source water quality			Quality of the potential water sources						
	Sun et al. (2012)	Schuh (2010)	EPA, 2016b	Drinking water	Fresh groundwater	Brackish groundwater	The Meuse at Keizersveer	The Rhine at Lobith	Wastewater (The Meuse watershed district)	Seawater
pH	6–8	6–8.5	6.5–8.1	7.4–8.7	5.0–8.0	6.5–7.5	7–8.2	7.7–8.1	n.d.	8.1
B, mg/l	< 15	n.d.	< 10	< 0.36	< 0.5	n.d.	< 0.08	< 0.09	n.d.	< 4.2
Cl, mg/l	n.d.	< 40000	< 90000	< 130	< 500	< 16300	< 62	< 115	< 3500	< 17765
Na, mg/l	n.d.	n.d.	< 5000	< 97	< 150	< 9200	< 46	< 67	< 110	< 9700
Fe, mg/l	< 20	< 10	< 15	< 0.06	< 130	< 9.7	< 2.1	< 1.9	< 0.25	< 1.52
Sr, mg/l	n.d.	< 5	< 1	n.d.	< 1	n.d.	< 0.2	< 0.6	< 0.25	< 13
Ba, mg/l	n.d.	< 5	< 38	< 0.1	< 0.2	n.d.	< 0.03	< 0.11	< 0.004	< 0.018
Si, mg/l	< 20	n.d.	< 20	< 10.7	n.d.	< 6.2	< 4.3	< 3.5	n.d.	< 0.42
Ca, mg/l	< 500	< 2000	< 4200	< 116	< 500	< 790	< 72	< 84	< 100	< 350
Mg, mg/l	n.d.	< 2000	< 1000	< 12	< 100	< 1080	< 9.2	< 13	< 14	< 1200
K, mg/l	n.d.	n.d.	< 500	< 8.5	< 25	< 330	< 9.4	< 5.9	< 54	< 350
PO ₄ , mg/l	n.d.	< 5	< 10	< 0.16	< 5	< 2.8	< 0.3	< 0.4	n.d.	< 0.5
SO ₄ , mg/l	< 50	< 500	< 1000	< 97	< 700	< 2290	< 68	< 84.7	< 356	< 2350
HCO ₃ , mg/l	< 1000	< 300	n.d.	< 340	< 500	< 424	< 190	< 200	n.d.	< 160
Bacteria, counts/ mL	n.d.	< 100	< 100	< 100 ^a	n.d.	n.d.	n.d.	< 24300 ^b	n.d.	n.d.

n.d. - not determined.

The water quality parameters, which exceed the recommended values for hydraulic fracturing operations are marked in bold.

^a Total bacterial colony numbers at 25°C on R2A agar plates.

^b Total bacterial colony numbers at 20°C on R2A agar plates.

disintegration of fracturing fluid components is site specific and depends on specific influent conditions and technical requirements (Cheremisinoff and Davletshin, 2015). As a rule, gel fluids require a higher quality of the source water than slick water fluids (Schuh, 2010). Requirements for source water for fracturing fluid formulation from different U.S. operators are listed in the Table 2 (EPA, 2016b; Schuh, 2010; Sun et al., 2012). Table 2 also lists the long-term quality parameters of drinking water, groundwater, river water, WWTP effluents, and sea water measured within the Dutch Posidonia shale (Rijksoverhead, 2016; Rijkswaterstaat, 2017; RIWA-Maas, 2016; RIWA-Rijn, 2016; Sjerps et al., 2017a; Stuyfzand and Raat, 2010) (see SI for the detailed description of the used datasets).

Drinking water quality complies with the requirements for fracturing fluid source water, except for the sulphate concentrations and upper pH limit, which exceed guideline values only at several locations (Table 2, Table S3). Therefore drinking water at most of the production locations may be directly used for fracturing fluid formulation.

Shallow fresh groundwater is often characterised by comparatively high iron and sulphate concentrations (Table 2). Both components can lead to scaling via iron oxides or barium sulphate formation. Iron removal is usually implemented in the drinking water treatment plants either by iron oxidation at the treatment plant or injecting limited volume of aerobic water to the subsurface formation (subsurface iron removal) (Mendizabal and Stuyfzand, 2009). Similar technologies may be applied to remove iron from the source water for frac fluid formulations. Sulphate concentrations in the shallow aquifers vary depending on the natural subsurface properties. Thus, sulphate concentrations exceeding 150 mg/l often occur in the surface groundwater in the western part of the Netherlands due to the presence of the sulphur-containing sediments of marine origin (Fraters and de Goffau, 2014). Sulphate concentrations in the groundwater of the Eastern part of the country are much lower and generally do not exceed 50 mg/l. In addition, local increase in sulphate concentrations in shallow aquifers might occur due to the excess utilization of sulphur-containing fertilizers on agricultural land (Fraters and de Goffau, 2014). Sulphate, when present in fracturing fluid, forms barium salts, if barium is present in the connate water of the formation. Barium sulphate has extremely low solubility (< 3 mg/l), thus scale formation can occur already at low sulphate and barium concentrations (Slutz et al., 2012). Several

technologies, including chemical precipitation, ion exchange and biological removal (Fernando et al., 2018) can be used for excess sulphate removal from groundwater.

Quality of the Meuse and the Rhine water also complies with the source water requirements for most of the parameters, except of sulphate concentrations and total bacteria counts. Sulphate concentrations only slightly exceed the lowest reported guideline value of 50 mg/l and are much lower than the guideline values presented by North Dakota State Water Commission (500 mg/l) and EPA (1000 mg/l) (Table 2, Table S4). In contrast, total bacterial counts in river water are two orders of magnitude higher than the source water guideline value of 100 cfu/ml. Therefore bacteria removal from river water is required to prevent bacterial fouling of the downhole equipment. The quality of the WWTP effluents in the Meuse watershed district regarding the selected parameters was comparable to the surface water quality (Table 2), which also implies necessity of bacterial removal prior to the use of WWTP effluents as source water.

Quality parameters of the sea water and brackish ground water in the areas along the North Sea coast exceed guideline values for sulphate (2.5–50 times) strontium (2.6–13 times), sodium (1.8–2 times) and calcium (1.2 times) (Table 2, Table S5). The latter cations may change the viscosity of the fracturing fluid and interfere with friction reducers (Esmailirad et al., 2016; Li et al., 2016). Therefore treatment strategies for source water conditioning from the North Sea water or brackish groundwater should include techniques for sulphate and divalent cations removal.

3.1.4. Choice of the water source

The analysis of the quantity and quality of the available water sources for fracturing fluid formulations presented in the previous sections has shown that the trade-off between the water availability and quality has to be met. Drinking water does seldom require any additional treatment, however due to the high treatment and distribution costs its price is often higher than the price of the other water sources (van der Zeijden et al., 2009). Complex procedures for obtaining groundwater abstraction permits and additional taxes for groundwater abstraction by the industries may become constraints for the use of fresh groundwater from shallow aquifers as source water for fracturing fluids (van der Zeijden et al., 2009). In addition, fresh water can cause

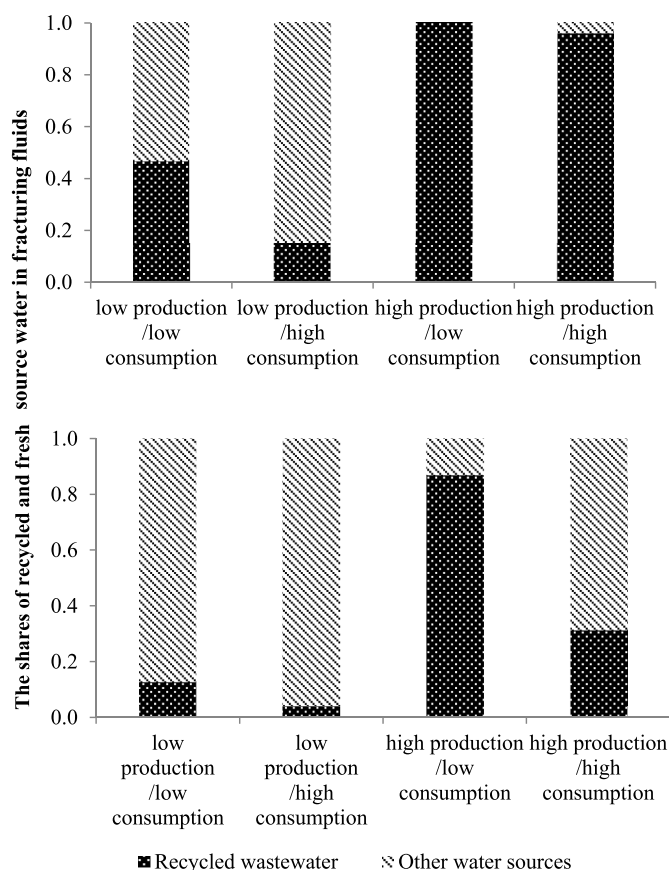


Fig. 3. The share of recycled wastewater as a source water for fracturing fluid formulations at 90% (A) and 25% (B) wastewater recycling.

expansion of the clay sediments of Posidonia formation due to its low ionic strength (Slutz et al., 2012). Sea water and brackish groundwater that are available without legislative restrictions require removal of a number of their natural constituents before the water can be used for fracturing formulations, which also incur high treatment costs.

River water having the highest availability after the sea water, has sufficient quality for fracturing fluid formulations, apart from the bacterial quality, which can be controlled by relatively cheap treatment methods, such as filtration or coagulation (Yang et al., 2014). Seasonal droughts could pose temporary limitations on the use of the Meuse as a primary water source (de Wit et al., 2007; RIWA-Maas, 2016). The average Rhine discharge is an order of magnitude higher than the Meuse discharge; moreover, it is less affected by the seasonal changes (RIWA-Rijn, 2016; Sjerps et al., 2017b). The projected maximal annual water use for fracturing (2.88 Mm³) constitutes only 0.03% of the annual water use of the Rhine water in the Netherlands (8598 Mm³) or 0.004% of the total renewable river water resources of the country (Graveland et al., 2017). Therefore river water is considered the most promising water source for hydraulic fracturing formulations. WWTP effluents are unlikely to be used directly, because many effluents contribute significantly to the water balance of small rivers and creeks in the region.

3.2. Estimated wastewater production

3.2.1. Estimated wastewater quantity

The estimated total wastewater production in the onshore Dutch Posidonia shale varies between 6.6 Mm³ (low water consumption and low wastewater recovery) and 48.0 Mm³ (high water consumption and high wastewater recovery). The maximal annual wastewater production of 0.48–3.49 Mm³ respectively is expected in the 16th year from

the beginning of the shale development (Table S2). Combination of high water consumption with low wastewater recovery will result in 19.8 Mm³ of total wastewater production and annual wastewater production of 1.44 Mm³ in the peak year. Low water consumption together with high wastewater recovery will result in a similar wastewater production (15.9 Mm³ and 1.15 Mm³ for total and annual wastewater production respectively). Wastewater production at high water consumption and high wastewater recovery (48.0 Mm³ or 39000 m³/well) is highly unlikely. According to Kondash et al. (2017) median wastewater volumes for major unconventional formations in the U.S. are ranging between 1720 and 14320 m³. The median long-term wastewater production from shale gas wells at Barnett formation is 11900 m³ according to Nicot et al. (2014). Therefore, wastewater production at the Dutch Posidonia shale is unlikely to exceed the value estimated for the combination of high water consumption and low wastewater recovery scenario (19.9 Mm³), which gives the median wastewater production of 16200 m³/well.

The volume of injected fluid is only one of the numerous factors, which influence wastewater production from shale gas wells. Therefore, estimation of wastewater production as a percentage of water consumption only gives indicative values, which can be significantly different depending on the numerous factors, including the formation pressure, interactions between the injected fluid and formation, and possible fracking of the more permeable reservoirs, such as sandstones and limestones, or connections with natural open faults (EPA, 2016a; Liu et al., 2015). Posidonia formation is overpressured and has low brittleness (Janzen, 2012). High formation pressure provides higher energy to flowback, increasing wastewater recovery. Low brittleness implies less developed fracturing network after fracturing event, which decreases hydration of shales and also increases wastewater recovery (Liu et al., 2015). On the other hand, high gas content predicted for the Posidonia shale means that the volumes of the connate water of the formation will be low, which should decrease wastewater production (Janzen, 2012). Combination of these factors and lack of exploration data for Posidonia shale makes impossible more precise estimation of the flowback and produced water volumes for shale gas production.

3.2.2. Impact of wastewater recycling on fresh water consumption and wastewater disposal

Recycling wastewater for fracturing of the new wells has become a widespread management option for flowback and produced water in the shale gas basins. Wastewater recycling can partially substitute demand for the fresh water for fracturing fluid formulations. The recycling rates in the U.S. vary greatly from > 90% recycling in Marcellus shale to zero recycling in the North Dakota (Chen and Carter, 2016). The spread of recycling as management option for shale gas wastewater depends on multiple factors, including the composition of flowback and produced water, scheduling of the new fracturing events, availability of storage and transportation facility, availability of technologies for recycled wastewater conditioning for fracturing fluid formulations etc. (Rahm et al., 2013; Rahm and Riha, 2014).

Ratio of recycled wastewater to consumed water in the Dutch Posidonia shale under all water consumption and wastewater production scenarios is shown in Fig. 3. The water demand for the new wells could be covered only with high wastewater production scenarios (> 110% of wastewater recovery) and 90% wastewater recycling (Fig. 3A). At 25% recycling shale gas wastewater will cover 4–86% of the total water demand depending on the combination of the wastewater production and water consumption scenarios (Fig. 3B). The coverage of source water demand also depends on the formation development over time (Fig. 4). Wastewater recycling will cover the higher share of the source water demand for fracturing fluid formulations in the later stages of the formation development, when higher amounts of flowback and produced water from multiple operation wells will be available.

Alternative management options for shale gas wastewater include

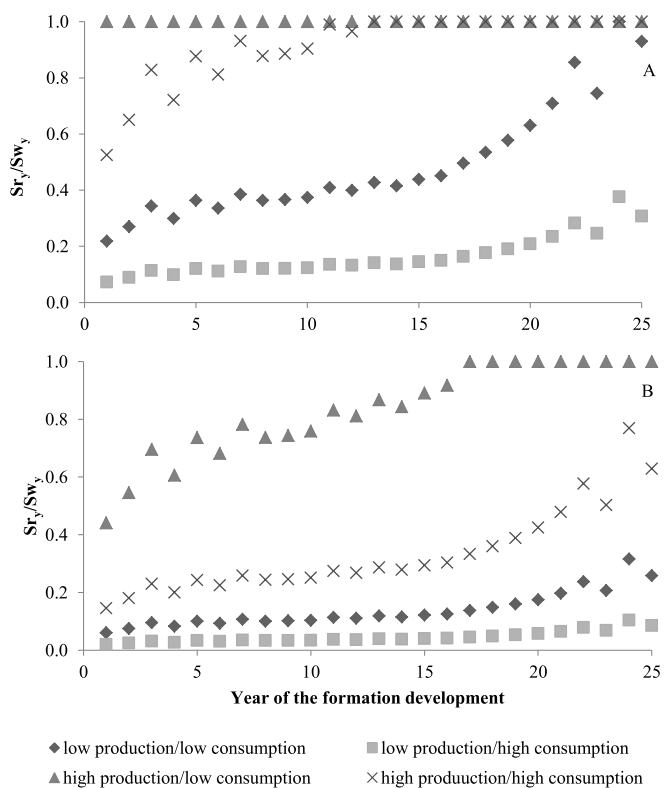


Fig. 4. Ratio of the annual recycled wastewater (S_{ry}) to the annual water consumption for fracturing fluid formulations (S_{wy}) at the different combinations of water consumption and wastewater production scenarios at 90% (A) and 25% (B) wastewater recycling.

disposal i) into the injection wells, ii) into the sea after removal of toxic components or iii) into surface waters after removal of toxic components and desalination (Rahm et al., 2013). Injection to the subsurface is used in the North of the Netherlands as a disposal strategy for brackish water, including produced water from the oil field (Benneworth and Velderman, 2016; Woltheek et al., 2013). It requires presence of the formations with suitable geological properties, e.g. high porosity and low permeability (Gregory et al., 2011). Empty oil and gas

fields are often suitable locations for shale gas wastewater disposal. However the main developed conventional gas field in the Netherlands, Groningen gas field, is located at the distance of 150–200 km from the Posidonia shale so costs for transportation will be too high to make this an attractive possibility. Disposal to the sea after treatment is the most feasible option for shale gas wastewater management, which requires removal of toxic wastewater components (organic matter and natural radioactive material) but does not require expensive desalination. Combination of biological and physic-chemical treatment is feasible for application and disposal of saline effluent after treatment of industrial wastewater is already applied at Delfzijl WWTP (The Netherlands) (Butkovskiy et al., 2018; van der Marel and de Boks, 2014). Disposal to the fresh water after treatment, though being a common disposal route for most of the municipal and industrial wastewater, will require application of expensive desalination technologies in case of the extremely saline flowback and produced waters. Moreover, brine, produced by the common desalination technologies, accounts for 30%–60% of the treated wastewater by volume and requires further treatment (Igunnu and Chen, 2014).

3.2.3. Estimated wastewater quality and wastewater recycling

Wastewater quality is the main constraint which hinders its recycling on a large scale by many operators in the U.S. Information on the flowback and produced water composition from shale gas plays in Europe is very scarce and is limited to the data from few hydraulically fractured wells in Poland and Germany (Kantor et al., 2015; Olsson et al., 2013). These data are compared to the quality required for fracturing fluids in Table 3. Average concentrations of Na, Ca, Mg, Fe, Ba and Sr in the flowback water from German exploration well (Damme 3) are 6.1, 3.4, 1.8, 9.1, 91.0 and 1455 times higher than the guideline values for fracturing fluid formulations. The same parameters in the flowback water from Polish exploration well also exceed guideline values.

Ba and Sr removal requires application of specific removal technologies, e.g. coagulation, precipitation or ion exchange (Cogan, 2016; Jiang et al., 2013). Concentrations of the other contaminants can be decreased to the accepted guideline values for wastewater recycling either applying (partial) distillation, or diluting shale gas wastewater with fresh water (Butkovskiy et al., 2017). Dilution of recycled shale gas wastewater by either a factor of ten or two will increase the volume of water which has to be disposed. According to these results,

Table 3 Recommended source water quality for hydraulic fracturing operations and flowback water composition from hydraulically fractured shale gas wells in Europe.

Parameter	Recommended source water quality			Flowback water composition from hydraulically fractured shale gas wells in Germany and Poland			
	Sun et al. (2012)	Schuh (2010)	EPA, 2016a,b	Damme 3 (Germany) ^a			Baltic shale gas basin (Poland) ^b
				Minimal	Maximal	Average	
pH	6–8	6–8.5	6.5–8.1	n.d.	n.d.	n.d.	7–8.2
B, mg/l	< 15	n.d.	< 10	n.d.	n.d.	n.d.	44
Cl, mg/l	n.d.	< 40000	< 90000	40360	88440	78229	54316
Na, mg/l	n.d.	n.d.	< 5000	17690	36390	30582	17509
Fe, mg/l	< 20	< 10	< 15	23	160	91	39
Sr, mg/l	n.d.	< 5	< 1	790	1720	1455	1235
Ba, mg/l	n.d.	< 5	< 38	180	593	455	198
Si, mg/l	< 20	n.d.	< 20	n.d.	n.d.	n.d.	n.d.
Ca, mg/l	< 500	< 2000	< 4200	6700	16500	14120	10201
Mg, mg/l	n.d.	< 2000	< 1000	890	2130	1799	1132
K, mg/l	n.d.	n.d.	< 500	52	157	110	1569
PO ₄ , mg/l	n.d.	< 5	< 10	n.d.	n.d.	n.d.	< 0.1
SO ₄ , mg/l	< 50	< 500	< 1000	4	15	8	10.8
HCO ₃ , mg/l	< 1000	< 300	n.d.	n.d.	n.d.	n.d.	n.d.
Bacteria, counts/mL	n.d.	< 10000	< 10000	n.d.	n.d.	n.d.	n.d.

n.d. - no data available.

^a n = 10.

^b Obtained under non-disclosure agreement regarding location of the well.

wastewater management will present a serious challenge for the development of the Posidonia shale gas field, because even at high recycling rate the majority of the produced wastewater has to be treated either for recycling or for disposal.

3.3. Strength and weaknesses of the applied methodology

Shale gas production is a highly debated topic regarding the balance between its economic merits and environmental impacts (Howarth et al., 2011). The shale gas formations with potential for commercial production are located in many parts of the world, including Europe. However, large-scale commercial shale gas production is so far limited to few countries, namely United States, Canada and China (EIA, 2015a). High costs for commencement of exploration wells, high heterogeneities within formations and legislative bans for shale gas exploration determine the lack of the data that are necessary for precise estimation of water-related environmental impacts in other parts of the world. At the same time, industry, legislation, water management and society sought to answer the questions regarding potential water-related impacts of shale gas production.

Among the countries with established shale gas production the United States is the only one with developed shale gas industry characterised by over 20 years of experience in horizontal drilling and hydraulic fracturing within several formations located in different geographical and geological conditions, extensively documented environmental issues and a vast number of scientific data on the water-related impacts of fracking (Wang et al., 2014). Contrary, shale gas production in Canada and China is limited to the few relatively small plays with scarce data on environmental impacts.

The methodology for rough estimation of water cycle related to shale gas production, which is based on the available macroscale data on water consumption and wastewater production from developed formations is proposed in this study and applied for the Dutch Posidonia shale. The main steps include:

- Estimation of the prospective area for shale gas production based on the data on surface land use and shale maturity, if the latter is available;
- Rough estimation of the lower and upper limits of the total and annual water consumption for fracturing fluid formulation, based on the available data and methodologies for the developed formations in the U.S.;
- Comparison of the estimated water volumes with the available water reserves on the regional and/or national level;
- Estimation of the total and annual wastewater volumes as a percentage of consumed water sources;
- Comparison of the estimated wastewater volumes with available disposal methods and potential for wastewater recycling.

The proposed methodology is based on the data related to the U.S. shale formations, which geological properties would be different from the properties of the European shales (Gény, 2010; Le, 2018). The estimations may be also biased by the changes in the surface land use patterns, inadequate estimations of the recoverable shale gas resources, and development of novel technologies, which may change the water use pattern for fracturing. However, introducing the best and the worst case scenarios for water consumption and wastewater production by taking into account large datasets related to the water management in the various U.S. shale plays with different geographical and geological conditions this methodology aims to approximate possible upper and lower limits of water consumption of the undeveloped and largely unexplored shale. The input data used for this estimation, such as WUI, well density, lateral density etc. are either average values or weighted medians of the largely heterogeneous datasets related to several shale gas basins, which ensure heterogeneity being taken into consideration. The proposed methodology can help to set up limits for predicted water

consumption and wastewater production at the shale plays with high data uncertainties providing wide margins to consider for the unknowns and interplay differences. The comparison of the estimated water and wastewater volumes with available water reserves and wastewater disposal options gives thus the first approximation on the water stress induced by the shale gas production in the studied area. Thus, in case of the Dutch Posidonia formation estimated annual water consumption for shale gas production is $2 \cdot 10^4$ lower than annual river discharge, which ensure sufficient water supply and minimal stress on the water reserves in the region. However, this difference may be much lower for the arid regions, where more precise estimations after exploration works are required.

4. Conclusions

Estimated total water consumption for development of the onshore Dutch Posidonia shale over the period of 25 years varies between 12.2 and 36.9 Mm³. Estimated total flowback and produced water volumes over the formation development period vary between 6.6 and 48.0 Mm³. The latter value is not expected to occur in practice, as median wastewater production for the existing shale gas formations are much lower.

River water appears to be the most feasible source for fracturing fluid formulation because of its abundance and quality. Wastewater recycling may significantly decrease the volumes of fresh water required for preparation of fracturing fluids, but does not eliminate the need for wastewater treatment and disposal. Proposed approach for estimation of the water consumption and wastewater production within the shale gas field prior to its development can also be applied for other shale gas fields with high data uncertainties.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jenvman.2018.10.066>.

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