

Potential Microbial Load Reductions Required to Meet Existing Freshwater Recreational Water Quality Standards for a Selection of Mid-century Environmental Change Scenarios

Rory Coffey¹ · Brian Benham² · Karen Kline² ·
Mary Leigh Wolfe² · Enda Cummins¹

Received: 12 June 2015 / Accepted: 8 October 2015 / Published online: 25 October 2015
© Springer International Publishing Switzerland 2015

Abstract Adapting to the impacts of environmental change (land use and climate), specifically on water quality, is one of the most pressing challenges for water policy globally. Currently, European Union and United States regulatory structures fail to explicitly account for the potential threats caused by environmental change to water quality in recreational freshwater. The objective of this study was to use watershed modelling to assess possible future load reduction measures and management strategies that may be required to meet existing microbial water quality standards in freshwater systems. The modelling work considers a suite of possible future environmental change scenarios for two watersheds in the west of Ireland and one watershed in Virginia, USA. Extensive reductions in microbial source loads (predominantly from point sources) for the Pigg (US) and Black (Ireland) watersheds were required to meet existing US Clean Water Act standards for the future scenarios simulated. Under the European Union Bathing Water Directive, all watersheds assessed were within the “excellent” classification for the various environmental change scenarios, indicating that this standard may be more achievable in future years. Outcomes of the work suggest that managing potential risks by adapting existing regulatory frameworks to explicitly plan for the effects of environmental change must become a critical component of watershed planning in the intermediate term. This will provide safe recreational water for use in society and protect public health.

Keywords Environmental change · Water quality · Standards · Freshwater

✉ Rory Coffey
rory.coffey@ucd.ie

¹ UCD School of Biosystems Engineering, Agriculture and Food Science Center, University College Dublin, Belfield, Dublin 4, Ireland

² Department of Biological Systems Engineering, Virginia Tech, Seitz Hall, Blacksburg, VA 24061, USA

1 Introduction

Although there is uncertainty about the exact nature, magnitude, and timing of future environmental changes (specifically climate and land use), it is certain that impacts will affect human health, natural ecosystems and the built environment (Ahmadi et al. 2014; Balbus et al. 2013; Coffey et al. 2014; Crossman et al. 2013; El-Fadel et al. 2012; Hofstra 2011; Palmer et al. 2008). Research suggests that the impact of environmental changes on water quality may be intensified by the effects of climate change (Hofstra 2011; Murdoch et al. 2000; Najjar et al. 2010; St. Laurent and Mazumder 2014). However, few studies have assessed the potential extent of these changes and their possible implications for environmental health (Hofstra 2011; Kundzewicz et al. 2008). Of particular concern is microbial contamination (containing potentially pathogenic microorganisms such as enterotoxigenic *Escherichia coli* and protozoan parasites e.g., *Cryptosporidium*), which is a major cause of surface water impairments globally (Coffey et al. 2007, 2014; Frey et al. 2013; Hofstra 2011; Oliver et al. 2014; Pandey et al. 2014). In the USA, microbial contamination of water sources is already a leading cause of surface water pollution (Pandey et al. 2014; USEPA 2004); while in Europe diffuse microbial pollution, in particular from agriculture, is reported to affect most surface water bodies (Kristensen 2012). Several studies detail that future changes in precipitation, temperature, land use, human population and agricultural production will cause increased transport of microbial contaminants (such as *E. coli*) to water sources (Coffey et al. 2014; Dunn et al. 2012; El-Fadel et al. 2012; Gray 2014; Hampson et al. 2010; Macleod et al. 2012; Vermeulen and Hofstra 2014). These environmental changes will impact waterborne microbial concentrations in surface water worldwide, possibly increasing the risk of diseases caused by human exposure to waterborne pathogens (Balbus et al. 2013; Boxall et al. 2009; Coffey et al. 2014; Hofstra 2011; Patz et al. 2005; Vermeulen and Hofstra 2014). As the infective dose for pathogenic microorganisms, such as enterotoxigenic *E. coli* and *Cryptosporidium*, is low, even slight environmental changes may result in higher incidence of waterborne illnesses (Coffey et al. 2014; Delpla et al. 2009; Gray 2014; National; Patz et al. 2005; Vermeulen and Hofstra 2014).

As drinking water abstracted from surface water sources is normally treated to remove/disinfect microbial pathogens prior to the distribution (EU 1998), a major risk is posed to public health from microbial contamination of freshwater due to recreational use (e.g., swimming, canoeing etc.) (Coffey et al. 2007; Semenza and Menne 2009; Soller et al. 2010). Within the USA, the US Environmental Protection Agency (US EPA) aims to protect ambient water quality in rivers and streams through criteria detailed in the Clean Water Act (CWA) and the implementation of the Total Maximum Daily Load (TMDL) program. The criteria aim to protect primary contact recreation. In Europe, the Bathing water directive (EU BWD-revised in 2006 to update the original 1976 Directive) aims to protect human and environmental health and complements the Water Framework Directive (WFD-where bathing waters are one of the protected areas) (Oliver et al. 2014). Microbial standards are provided only for designated bathing areas of coastal and freshwater, and little focus has been concentrated on microbial contaminants in most watersheds (unless designated as a bathing location). The revised directive will be fully implemented in 2015 and incorporates bathing water standards specifically for inland freshwater, and also includes more stringent standards for intestinal enterococci and *E. coli* (EU 1998, 2006a). Failure to comply with the WFD and ensure good water quality for freshwater that drains into bathing waters could result in failure to comply with the BWD. The revised directive was introduced to ensure compatibility with the WFD and to incorporate scientific advances (Kay et al. 2007).

For freshwater, fecal coliforms and *E. coli* are considered to be a good surrogate for enteric bacterial pathogens and protozoan parasites (Coffey et al. 2007). Therefore, fecal coliforms and more recently *E. coli* are commonly used organisms as an indicator of the general microbiological quality of water, as these microorganisms are always present in fecal contamination from human and animal sources (Coffey et al. 2007). Regulatory bodies have historically based fecal coliform/*E. coli* criteria for recreational water quality upon either percentage compliance levels, typically 95 % compliance levels (i.e., 95 % of the sample measurements taken must lie below a specific value in order to meet the standard), or geometric mean values of water quality data collected from the water body over some specified period of time (Ahmadi et al. 2014). US criteria are based on the geometric mean, whilst the EU standards follow a 95th percentile approach (Kay et al. 2004). A summary of standards for microbial contamination in recreational freshwaters for *E. coli* established in the US and EU is given in Table 1.

Current studies on the fate and transport of microbial contaminants have detailed that these existing microbial quality standards are sufficient to protect water sources and public health in the long term (Coffey et al. 2014); however, given projected environmental changes, meeting these standards will be challenging for those tasked with environmental management (including stakeholders and regulators) and projects aimed at meeting existing water quality standards (Coffey et al. 2014; Wilby et al. 2006a). More restrictive watershed management measures are likely to be required to meet standards, meaning that stakeholders, regulators and policy makers may need to review and update previous water quality remediation actions for robustness to environmental change. To date, EU and US regulatory structures fail to explicitly account for the potential threats to water quality and public health posed by environmental change (Coffey et al. 2014; Quevauviller 2011; Wilby et al. 2006a).

The US TMDL program (as defined in the CWA) aims to establish ambient water-quality based limits on the allowable pollutant load for impaired water bodies that fail to meet appropriate water quality standards (Copeland 1999). Typically, coupled watershed-scale hydrology and water quality models are used to relate pollutant loads from different point and nonpoint sources to in-stream water quality, and to develop scenarios for achieving the TMDL pollutant load target through modeled reductions in the identified pollutant sources (Radcliffe et al. 2009). In the EU, the WFD places greater emphasis on the desired ecological status as opposed to the pollutant load targets. To check compliance with water quality

Table 1 US and EU water quality standards established for *E. coli* in recreational freshwaters

Region	Indicator	Guideline value	References
United States	<i>E. coli</i>	^a Geometric mean concentration: ≤ 126 CFU 100 mL ⁻¹ ^b Single sample maximum concentration: 235 CFU 100 mL ⁻¹	US EPA (2012)
European Union	<i>E. coli</i>	<500 CFU 100 mL ⁻¹ (excellent) ^c <1000 CFU 100 mL ⁻¹ (good) ^c <900 CFU 100 mL ⁻¹ (sufficient) ^d	EU (2006)

CFU colony forming units

^a 0 % violation rate

^b <10.5 % violation rate

^c Based upon 95th percentile evaluation

^d Based upon 90th percentile evaluation

standards, the WFD requires the establishment of monitoring programmes, with the minimum frequency typically set at monthly (Gevaert et al. 2009). The use of water quality models in this instance has focused more on research and the identification of critical source areas, rather than the development of water quality assessment and restoration planning. Due to the lack of consistent monitoring approaches and the demanding timeframe of the WFD, the use of combined monitoring and watershed modeling applications to link changes in pollutant sources and water quality with the effects of meteorological drivers is more frequently being suggested (Coffey et al. 2007; Collins and McGonigle 2008; Gevaert et al. 2009).

Use of the same integrated hydrologic and water quality models has also been recommended as a viable means of assessing future impacts of environmental change on water quality and hydrologic response (Boxall et al. 2009; Coffey et al. 2014; Delpla et al. 2009; Hofstra 2011; Johnson and Weaver 2009; Larsen et al. 2013; Wilby et al. 2006b). When integrated with monitoring, models can assist in understanding how climate and land management affect hydrology and water quality at the watershed scale (Imhoff et al. 2007; Johnson et al. 2011; Larsen et al. 2013). Their use can be essential to examine trends of contaminants in the environment and to forecast future conditions and scenarios. Capabilities can also be extended to explore effective best management strategies that could be included in policy planning to help ensure that future water quality standards continue to be met (Coffey et al. 2014). At present, it is difficult to forecast how microbial transport will evolve under future climate and watershed conditions. However, as climate change is to an extent uncontrollable, it is likely that more restrictive control measures to reduce microbial source loads will be required to meet regulatory standards and maintain good ecological condition for freshwaters. Adapting to climate change, in particular for water-related impacts, remains one of the most difficult challenges for global water policy in future years (Quevauviller 2011). Hence, the objective of this study was to use watershed modelling to assess the possible extent of future microbial load reductions that may be required to meet existing freshwater microbial water quality standards (US and EU). The modelling work considers a suite of possible future environmental change scenarios for watersheds in the west of Ireland and Virginia, USA.

2 Methods

2.1 Study Areas

2.1.1 Ireland

The Fergus watershed (518 km²) and the Black watershed (193 km²) are located in the west of Ireland (Fig. 1a). Agriculture is the dominant land use in both watersheds, with substantial areas of pasture (>50 %). Manure applied to land as fertiliser, manure deposited from grazing livestock and septic systems are the major diffuse fecal pollution sources. Point sources of fecal pollution include wastewater treatment plants (3) and direct stream deposits from livestock (Black watershed only). Both study locations are karst in nature, i.e., are underlain by limestone rock. Annual rainfall ranges between 1000 and 1600 mm. The mean daily temperature ranges from 9.3 °C to 10.7 °C. The Black River drains to Lough Corrib. Although not designated as a bathing location, the lake is commonly used for recreational activity. The Fergus River drains to the Shannon estuary, which has one designated bathing area. The river also drains Ballyallia Lake, which is a designated bathing area.

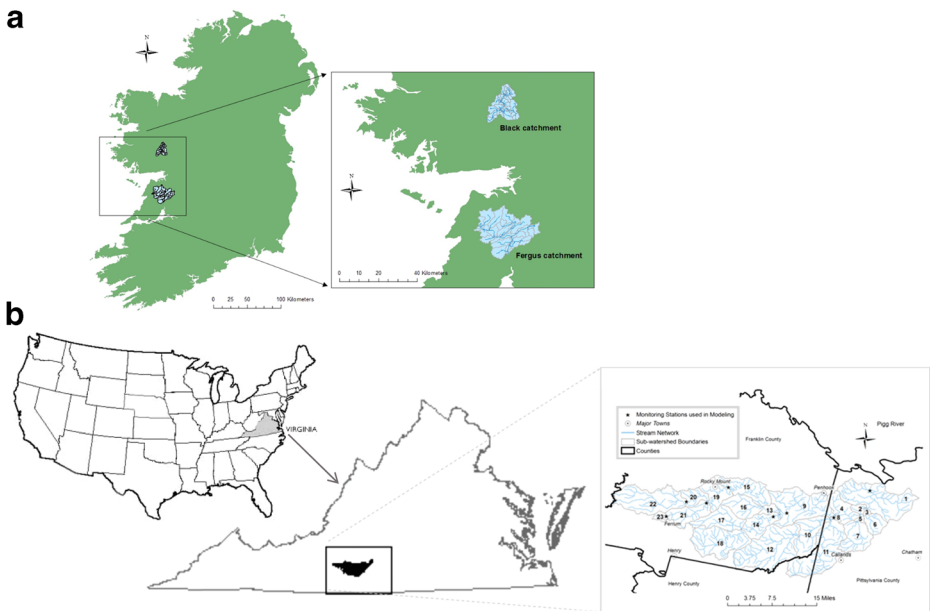


Fig. 1 a Location of the Fergus and Black watersheds in Ireland and b Location of Pigg river watershed in Virginia, USA

2.1.2 USA

Pigg River is a tributary of the Roanoke River in southwestern Virginia with a drainage area of 1015 km² (Fig. 1b). Story Creek, Snow Creek and Big Chestnut creek are the main tributaries of Pigg River. The watershed consists primarily of forest (72 %) and agricultural land (26 %). Livestock, manure application, wildlife, pets and failing septic systems are the main sources of diffuse fecal pollution (transported to the water channel *via* surface runoff). Key point sources of fecal pollution (deposited directly to streams) in the watershed include wastewater treatment plants (2), straight pipes from residential housing (14), and direct deposits from cattle and wildlife defecating in streams. Long-term climate records for the area show an average annual precipitation of 1132 mm, with 54 % of the precipitation occurring from May to October. Average annual snowfall at Rocky Mount (located within the watershed) is 424 mm, with the highest snowfall occurring during January. Average annual daily temperature is 13.1 °C. The highest average daily temperature of 24 °C occurs in July, while the lowest average daily temperature of 2.1 °C occurs in January.

2.2 Modelling Frameworks

2.2.1 Hydrology and Water Quality

This analysis is based on a set of modelling frameworks that were previously developed to simulate hydrology and microbial transport for the study watersheds (Benham et al. 2006a; Coffey et al. 2010; Coffey et al. 2013). The Hydrological Simulation in Fortran (HSPF) (Bicknell et al. 1997) was used to simulate end points for the Pigg River watershed. For the Fergus and Black watersheds, model simulations were conducted using the Soil and Water

Assessment Tool (SWAT) (Winchell et al. 2007). Watershed specific data on climate, microbial sources, soils, land use/management, and topography were used in the initial development of the modelling frameworks. Observed hydrological data from watershed located flow gauges, and observed *E. coli* (Fergus and Black) and fecal coliform (Pigg) sampling data were used to calibrate and validate individual models.

The coefficient of determination (R^2) and Nash Sutcliffe Efficiency (NSE) statistical metrics are used to quantify the agreement between the observed and simulated model outputs for hydrology (Moriassi et al. 2007). For the Fergus, Black and Pigg catchments, R^2 and NSE metrics for daily average stream flow are provided in Table 2. Contemporary monitoring of fecal indicator bacteria is time, labor and cost intensive. Thus, in-stream bacteria concentrations are typically sampled infrequently (on a monthly basis, at best). As these samples represent only an instant in time, it is not reasonable to expect any model to accurately simulate a daily average concentration equal to an observed value on a particular day (Kim et al. 2007). For the Pigg river watershed, a temporal-window statistic is used for simulated hourly-concentrations of fecal coliform over a period of 5 days to calculate the minimum-maximum range that is compared to observed in-stream bacteria concentrations (Kim et al. 2007). For the Fergus and Black watershed (simulated on a daily time step), combinations of metrics detailed by Kim et al. (2007) and Moriassi et al. (2007) are used to compare observed and simulated in-stream daily *E. coli* concentrations. Results for all three study watersheds are given in Table 3. Despite limited data availability and associated uncertainties, it was concluded that the calibrations achieved were acceptable to examine the sensitivity of microbial contamination, identify high risk periods and predict potential microbial loads (Coffey et al. 2010, 2013).

2.2.2 Environmental Change Scenarios

Input data for the hydrology and water quality modeling frameworks for all three watersheds were updated using readily accessible future climate and land management data projections for the mid-century period (circa 2050). For Pigg River, one future climate scenario (CC) and one future land management scenario (LU) were considered and used to simulate impacts on daily stream flow and microbial load. Downscaled General Circulation Model (GCM) projections (ensemble of 7 GCMs) by the Consortium for Atlantic regional Assessment (CARA) for the B2 scenario (moderate greenhouse gas emissions) (IPCC 2001) were used to modify observed seasonal precipitation and temperature data (relative change) at Rocky Mount. A land management scenario (LU) reflecting available data on agricultural production and human

Table 2 Statistical metrics for the calibration and validation of daily average stream flow (m^2s^{-1}) for the Fergus, Black and Pigg rivers

Watershed	Calibration		Validation	
	^a NSE	^b R^2	NSE	R^2
Fergus	0.82	0.89	0.90	0.86
Black	0.85	0.85	0.77	0.72
Pigg	0.44	0.46	0.57	0.57

^a Nash Sutcliffe Efficiency

^b Coefficient of Determination

Table 3 Comparison of in-stream bacteria model calibration and validation statistics for the Pigg, Fergus and Black watersheds

Watershed (organism)	Criteria	Pigg (fecal coliform)		Fergus (<i>E. coli</i>)		Black (<i>E. coli</i>)	
		Calibration	Validation	Calibration	Validation	Calibration	Validation
Daily assessment							
^a PBIAS	±100 %	-71 %	-70 %	42 %	65 %	38 %	58 %
Median	±100 %	-23 %	-1 %	-11 %	11 %	-12 %	71 %
Geometric mean	±100 %	-47 %	1 %	41 %	5 %	36 %	76 %
Hourly assessment							
% of observed (obs) values within 5 day min-max range	>70 %	48 %	45 %	-	-	-	-
% of obs values >5 day max	~equal to % of obs values <5-day min	38 %	35 %	-	-	-	-
% of obs values >5 day min	~equal to % of obs values >5-day max	14 %	21 %	-	-	-	-

^a Kim et al. (2007) recommend ±100 % for bacteria; Moriasi et al. (2007) recommend ±70 % for nutrients (N and P)

population changes was used to update microbial source input data. For the Fergus and Black watersheds, two future climate scenarios [Representative Concentration Pathway (RCP) 4.5 and RCP 8.5) and one worst case LU scenario were considered in simulations to assess environmental change impacts on stream flow and microbial load. A downscaled multi-model ensemble (20 GCMs) (Gleeson et al. 2013) provided local scale projections for the RCP 4.5 (medium-low greenhouse gas emission scenario) and 8.5 (high greenhouse gas emission scenario) scenarios. Forecasted seasonal variations in temperature and precipitation were used to modify baseline observed weather data. A LU scenario, reflecting forecasted changes in human population (CSO 2013; Eurostat 2014) and a realistic increase in agricultural production (i.e., potential worst case in terms of microbial source load), was used to update microbial input data for both Irish watersheds.

Combinations of CC scenarios (Pigg: B2 only; Fergus and Black: RCPs 4.5/8.5) and the LU scenarios were subsequently simulated to examine changes in hydrology and microbial load. All analyses were based on simulation results expressed as mid-21st century changes relative to historical baseline conditions (the model calibration and validation periods). Resulting outputs were also evaluated under different flow conditions (i.e., low, medium, high) in each watershed. A comprehensive assessment of simulated changes in stream flow and microbial load under different flow conditions and environmental change scenarios are given in Coffey et al. (2015a) and (2015b). Overall, results indicated that microbial loads would increase under the various LU and CC scenarios simulated (Coffey et al. 2015a, b). Trends in microbial load changes were symptomatic of projected seasonal precipitation patterns, with the main increases evident in winter and fall. Increases in microbial sources from LU changes (increase in human population and agricultural production), in combination with forecasted climate change, caused the most significant increase in daily microbial load for all watersheds (Coffey et al. 2015a, b). High and low flow conditions represented the periods when the greatest increase in microbial load was evident in all watersheds. The analyses indicated that future variations in land use/management may be as important as the effects of climate change on in-stream microbial pollutant loads. Results from all three watersheds, suggested the need to increase land management strategies that would reduce microbial source loads, and develop further understanding what watershed characteristics may be controllable in the future (Coffey et al. 2015a, b).

2.3 Simulation of Microbial Load Reductions

The main objective of the current study was to examine likely microbial source reductions that would be required to meet existing US and EU recreational water quality standards under future environmental change. Initially, scenarios previously simulated (Section 2.2) for each watershed were assessed to determine violations of the US CWA and EU BWD standards. For scenarios that violated these standards, microbial source load reductions were subsequently simulated and estimated to comply with recommended guideline values.

2.3.1 US CWA Assessment

Section 303(d) of the US CWA and the US EPA Water Quality Planning and Management Regulations (40 CFR Part 130) require states to identify water bodies that violate state water quality standards and to develop TMDLs for such water bodies (Benham et al. 2006b). A TMDL establishes the maximum allowable pollutant loading from both point and nonpoint

sources for a water body, allocates the load among the pollutant contributors, and provides a framework for taking actions to restore water quality (Copeland 2003). The main objective for bacteria TMDLs is to determine the reductions in *E. coli* loadings from point and nonpoint sources that are required to meet state/federal water quality standards (Benham et al. 2006a). The Virginia state water quality standards for *E. coli* used in the development of the TMDLs are 126 colony forming units (CFU) 100 mL⁻¹ (calendar-month geometric mean: 0 % violation rate) and 235 CFU 100 mL⁻¹ (single sample maximum: <10.5 % violation rate). During any assessment period, if more than 10 % of samples exceed the applicable standard, the stream segment associated with that sampling location is deemed impaired. The bacteria TMDL for any segments designated as impaired is developed to meet geometric mean standard.

When developing bacteria TMDLs, the required bacteria load reductions are modeled by decreasing the amount of bacteria applied to the land surface or directly to the stream. In this study, potential reductions allocated to meet the US CWA water quality standard for the various environmental change scenarios (detailed in Section 2.2) simulated for the three watersheds are compared. The goal was to achieve the following conditions:

1. Reduce bacteria loading from sources to meet a calendar month geometric mean concentration less than 126 CFU 100 mL⁻¹.
2. Reduce the bacteria loadings from controllable sources (excluding wildlife) such that violations of the instantaneous criterion (235 CFU 100 mL⁻¹) are less than 10.5 %.

Simulated reductions addressed both diffuse and point sources of microbial contamination. Since model calibration for the Pigg River watershed was conducted with fecal coliform inputs, a translator equation was used to convert the output to *E. coli*. The translator equation was developed by Virginia Department of Environmental Quality and converts fecal coliform model output to *E. coli* within the HSPF model. This conversion allows model comparison with the water quality standards (Benham et al. 2006b). The equation used was:

$$\text{Log}^2 E. coli (\text{CFU } 100 \text{ mL}^{-1}) = -0.0172 + 0.91905 \times \text{Log}^2 \text{ Fecal coliform } (\text{CFU } 100 \text{ mL}^{-1})$$

Annual loads were obtained by summing the daily loads and dividing by the number of years in the allocation period.

2.3.2 EU BWD Assessment

The EU BWD requires relevant authorities within EU member states to monitor the water quality of identified bathing areas throughout the bathing season (EU 2006b). In Ireland, the bathing season extends from June through September. Starting in 2015, three bathing water classifications are used: ‘sufficient’, ‘good’, or ‘excellent’. The classification standards are based on either a 90th or a 95th percentile value (Ahmadi et al. 2014). The concentration limits of *E. coli* permitted for inland freshwater are two times greater than those for coastal and transitional (marine) waters across all three classification standards.

The ‘sufficient’ standard is based on a lower percentile value, 90 %, compared with 95 % for ‘excellent’ and ‘good’, allowing for greater variance in the sample quality. In this study, the percentile standards for freshwater were used to ascertain the current and potential future

bathing status of the Pigg, Fergus and Black watersheds considering simulated water quality scenarios for *E. coli* concentrations. All environmental change scenarios (Section 2.2) were assessed to determine the possible future bathing status under the EU BWD classifications. The equations and steps used to calculate the percentile standards (EU 2006b) were as follows :

- (i) Calculate \log_{10} value of all bacterial concentrations in the data sequence to be evaluated
- (ii) Calculate the arithmetic mean " μ " of the \log_{10} values
- (iii) Calculate the standard deviation " σ " of the \log_{10} values
- (iv) Derive : Upper 90th percentile = $\text{antilog}(\mu + 1.282 \sigma)$
Upper 95th percentile = $\text{antilog}(\mu + 1.282 \sigma)$.

Using these steps, the percentiles during the bathing season (June through September) for each watershed were determined for the baseline and environmental change scenarios. If the resulting quantitative assessments for the various scenarios failed to meet the "good" bathing quality classification (i.e., 95th percentile $< 1000 \text{ CFU } 100 \text{ mL}^{-1}$), load reductions from key diffuse and point sources were considered until the appropriate percentile was achieved (similar to US TMDL reductions).

3 Results

3.1 US CWA Water Quality Assessment

Initially, outputs from all three watersheds were assessed to establish if simulated scenarios were violating the calendar month geometric mean standard (requires 0 % violation of $126 \text{ CFU } 100 \text{ mL}^{-1}$ for *E. coli*) and instantaneous standard (requires < 10.5 % violation rate of $235 \text{ CFU } 100 \text{ mL}^{-1}$ for a single sample). Table 4 details the results of this assessment for each watershed. For Pigg River, the geometric mean and instantaneous standard were violated under existing conditions and future environmental change scenarios. The Black watershed violated only the geometric mean standard. As a result, a reduction in source loads for the Pigg and Black watersheds were required to meet existing US microbial water quality standard under the various scenarios. Simulated concentrations of *E. coli* for the Fergus River met the US calendar month geometric mean standard and the instantaneous standard for all scenarios.

3.1.1 Allocation Scenarios

Pigg River A TMDL study was previously carried out for Pigg River, Snow Creek and Story Creek (all stream segments in Pigg watershed) in 2006 as they did not support the recreational (primary contact) designated use due to violations of the bacteria criteria (Benham et al. 2006b). Thus, source reductions to meet the calendar month geometric mean standard and the 10.5 % violation rate of the instantaneous standard for existing watershed conditions were previously simulated (as part of the TMDL study for each of the stream segments listed as impaired (Benham et al. 2006b). Initially, these source reductions were assessed to determine if they would suffice under future climate and land management scenarios (i.e., CC, LU, LU + CC). The reductions for the TMDL in 2006 did not result in a successful simulation for any of the environmental change scenarios. Subsequently, further reductions were applied to sources

Table 4 Violation rate of US CWA microbial water quality standards for the Pigg, Fergus and Black watersheds under existing conditions (baseline) and environmental change scenarios

	^a GEO	^b INS	Source reductions
Black			
Baseline	2 %	9 %	Yes
RCP 4.5	2 %	8 %	Yes
RCP 8.5	2 %	9 %	Yes
LU	8 %	10 %	Yes
RCP 4.5 + LU	8 %	9 %	Yes
RCP 8.5 + LU	8 %	10 %	Yes
Fergus			
Baseline	0 %	5 %	No
RCP 4.5	0 %	5 %	No
RCP 8.5	0 %	5 %	No
LU	0 %	6 %	No
RCP 4.5 + LU	0 %	5 %	No
RCP 8.5 + LU	0 %	6 %	No
Pigg			
Baseline	43 %	15 %	Yes
CC	47 %	24 %	Yes
LU	45 %	20 %	Yes
LU + CC	49 %	28 %	Yes

^a126 CFU 100 mL⁻¹ calendar month geometric mean standard (0 % violation rate)

^b235 CFU 100 mL⁻¹ instantaneous standard (<10.5 % violation rate)

to meet the TMDL goal of a calendar-month geometric mean concentration less than 126 CFU 100 mL⁻¹ and a single-sample maximum concentration of less than 235 CFU 100 mL⁻¹.

A summary of the reductions required to meet the geometric mean standard for Pigg River and other impaired segments (Snow Creek and Story Creek) is given in Table 5. For the TMDL study conducted in 2006, reductions in loading from cattle direct stream deposits (60–100 %), residential straight pipes (100 %), pasture land (95 %), and residential land (75–95 %) were required to meet the calendar month geometric mean standard for Pigg River, Snow Creek and Story Creek. The same sources were used as the focus of additional reductions required for the future environmental change scenarios. For the CC scenario, additional reductions were required for cattle direct stream deposits (Snow Creek: 65 %). Further reductions in wildlife direct stream deposits were necessary for Lower Pigg (45 %), Mid Pigg (20 %) and Story Creek (55 %) under the CC scenario. The LU scenario required higher load reductions for cattle direct stream deposits at Snow Creek (70 %), and wildlife direct stream deposits (50 %) at Story Creek to meet the geometric mean standard. When the combined impacts of the climate and land use scenario (LU + CC) were assessed, more severe load reductions from cattle direct deposits (70–100 %) and wildlife direct deposits (0–55 %) were necessary.

A summary of the reductions to meet the single sample standard violation rate is given in Table 6. The TMDL study conducted in 2006 recommended reductions in cattle direct deposits (10–100 %), straight pipes (100 %) and a reduction in sources from pasture for Lower Pigg and Snow Creek (both 95 %) for this standard. Under the future CC scenario, further reductions were required from cattle direct stream deposits (15–100 %) and some pasture land (Story Creek: 50 %). Greater reductions in loading from cattle direct deposits were only necessary to meet the standard under the future LU scenario. The combined climate and land

Table 5 Reductions required to meet the calendar month geometric mean standard (*E. coli*: 126 CFU mL⁻¹) for Pigg River and tributaries under existing conditions and future environmental change scenarios

	% Reduce cattle DD ^e	% Reduce cropland	% Reduce pasture	% Reduce loafing lot	% Reduce wildlife DD ^e	% Reduce straight pipes	% Reduce residential	% Reduce forest
Lower Pigg:								
TMDL (2006) ^a	100	0	95	0	30	100	90	0
CC ^b	100	0	95	0	45	100	90	0
LU ^c	100	0	95	0	30	100	90	0
LU + CC ^d	100	0	95	0	45	100	90	0
Snow Creek:								
TMDL (2006) ^a	60	0	95	0	0	100	95	0
CC ^b	65	0	95	0	0	100	95	0
LU ^c	70	0	95	0	0	100	95	0
LU + CC ^d	70	0	95	0	0	100	95	0
Upper Pigg:								
TMDL (2006) ^a	100	0	85	0	45	100	75	0
CC ^b	100	0	95	0	55	100	75	0
LU ^c	100	0	95	0	50	100	75	0
LU + CC ^d	100	0	95	0	55	100	75	0
Story Creek:								
TMDL (2006) ^a	100	0	85	0	45	100	75	0
CC ^b	100	0	95	0	55	100	75	0
LU ^c	100	0	95	0	50	100	75	0
LU + CC ^d	100	0	95	0	55	100	75	0

^a Reductions required to meet the standard for a Total Maximum Daily Load study conducted in 2006 (Benham et al. 2006a, b)

^b Climate change scenario: B2 emissions storyline

^c Land use scenario

^d Combined climate change and land use scenarios

^e Direct Deposits

Table 6 Reductions required to meet the single sample standard (*E. coli*: 235 CFU mL⁻¹) instantaneous violation rate (10.5 %) for Pigg River and tributaries under existing conditions and future environmental change scenarios

	% Reduce cattle DD ^e	% Reduce cropland	% Reduce pasture	% Reduce loafing lot	% Reduce wildlife DD ^e	% Reduce straight pipes	% Reduce residential	% Reduce forest
Lower Pigg								
TMDL (2006) ^a	10	0	0	0	0	100	0	0
CC ^b	55	0	0	0	0	100	0	0
LU ^c	40	0	0	0	0	100	0	0
LU + CC ^d	70	0	0	0	0	100	0	0
Snow Creek								
TMDL (2006) ^a	5	0	0	0	0	100	0	0
CC ^b	15	0	0	0	0	100	0	0
LU ^c	25	0	0	0	0	100	0	0
LU + CC ^d	35	0	0	0	0	100	0	0
Upper Pigg								
TMDL (2006) ^a	65	0	0	0	0	100	0	0
CC ^b	70	0	0	0	0	100	0	0
LU ^c	75	0	0	0	0	100	0	0
LU + CC ^d	85	0	0	0	0	100	0	0
Story Creek								
TMDL (2006) ^a	100	0	0	0	0	100	0	0
CC ^b	100	0	50	0	0	100	0	0
LU ^c	95	0	0	0	0	100	0	0
LU + CC ^d	100	0	50	0	0	100	0	0

^a Reductions required to meet the standard for a Total Maximum Daily Load study conducted in 2006 (Benham et al. 2006a, b)

^b Climate change scenario: B2 emissions storyline

^c Land use scenario

^d Combined climate change and land use scenarios

^e Direct Deposits

management scenario (LU + CC) required the largest reduction in cattle direct deposits across the watershed (35–100 %) for the single sample standard violation rate.

Black River Cattle direct deposits and application of manure were considered to be the main microbial sources in the watershed (Coffey et al. 2013). Allocated reductions in microbial loads from these sources were subsequently simulated to meet the US geometric mean standard. A 50 % reduction in load from cattle direct deposits and a 50 % reduction in loading from manure application were required for the baseline scenario (existing conditions). The same sources were also the focus for allocations simulated under the environmental change scenarios (climate and land use). For both CC scenarios (RCP 4.5 and 8.5) further reduction of load from cattle direct deposits (75 %) was sufficient to address the calendar month geometric mean standard. A reduction of 80 % for cattle direct deposits was required for the LU scenario. Complete exclusion of cattle from streams (100 % reduction) was necessary for combinations of both CC scenarios and the LU scenario. A summary of the required reductions allocated under various scenarios is given in Table 7. No allocated reduction in microbial sources were needed to meet the instantaneous violation rate (10.5 %) for the single sample standard (*E. coli*: 235 CFU mL⁻¹).

Fergus River No reductions were necessary to meet US CWA recreational freshwater standards.

3.2 EU BWD Water Quality Assessment

Simulated daily concentrations of *E. coli* during the bathing season (June through September) were assessed to determine bathing water status for each watershed. The results of this assessment are given in Table 8. All watersheds achieved the status of “excellent” under the EU BWD for freshwater under existing conditions. Although the 95th and 90th percentile increased under the various environmental change scenarios for each watershed, the bathing

Table 7 Reductions required to meet the calendar month geometric mean standard (*E. coli*: 126 CFU mL⁻¹) for the Black watershed under existing conditions (baseline) and future environmental change scenarios

Black	% Reduce cattle DD ^e	% Reduce manure application	% Reduce pasture	% Reduce septic systems
Baseline ^a	50	50	0	0
RCP 4.5 ^b	75	50	0	0
RCP 8.5 ^c	75	50	0	0
LU ^d	80	50	0	0
RCP 4.5 + LU	100	50	0	0
RCP 8.5 + LU	100	50	0	0

^a Baseline: calibrated and validated model based on observed input data for stream flow and *E. coli*

^b RCP 4.5: A medium-low greenhouse gas emission climate scenario

^c RCP 8.5: A high greenhouse gas climate scenario

^d LU: A realist worst case land use scenario

^e DD: Direct stream deposits

Table 8 Status of water quality for the Pigg, Fergus and Black watersheds determined using the EU BWD standards for inland waters

	CFU 100 mL ⁻¹		Bathing status ^a
	95th %	90th %	
Black			
Baseline	253	181	Excellent
RCP 4.5	258	188	Excellent
RCP 8.5	263	191	Excellent
LU	308	217	Excellent
RCP 4.5 + LU	319	227	Excellent
RCP 8.5 + LU	328	232	Excellent
Fergus			
Baseline	167	118	Excellent
RCP 4.5	163	118	Excellent
RCP 8.5	162	117	Excellent
LU	192	137	Excellent
RCP 4.5 + LU	187	188	Excellent
RCP 8.5 + LU	189	136	Excellent
Pigg			
Baseline	428	364	Excellent
CC	401	346	Excellent
LU	497	416	Excellent
LU + CC	491	418	Excellent

^aStatus based on classifications given in Table 1

status was still considered to be “excellent” based on the guideline values. Subsequently, no allocation scenarios were necessary to achieve the bathing standards set out in the EU BWD. The Pigg River watershed had the highest percentile ranges for the various scenarios simulated (95th%: 401–497; 90th%: 346–364).

4 Discussion

Current peer reviewed literature suggests that there will be changes in microbial water quality in future years (Coffey et al. 2014; Hofstra 2011; Vermeulen and Hofstra 2014). For all watersheds assessed in this study, increases in microbial loads were apparent under all environmental change scenarios simulated. Seasonal variations in the fate and transport of bacteria were proportionate to projected climate changes. However, when considering the results it should be noted that there are many uncertainties involved in the development of models and the simulation of environmental change scenarios. Sources of uncertainty could include sampling errors in observed data, data input errors (microbial source estimates, climate data-existing and downscaled), temporal and spatial variability, failure to capture in-stream processes, selection of climate scenarios and others (Benham et al. 2006a; Coffey et al. 2010, 2014; Dunn et al. 2012; Johnson et al. 2011; Johnson and Weaver 2009; Kim et al. 2010; Pandey et al. 2012; Piorkowski et al. 2014; Russo et al. 2011). For example, under typical settings Harmel et al. (2006) estimated cumulative probable error ranges associated with water quality data collection alone to be 6 to 19 % for streamflow, 7 to 53 % for sediment, and 8 to 110 % for N and P (bacteria are

likely to be in similar ranges to N and P). This emphasizes that many of the uncertainties cannot be accounted for in modelling frameworks and are considered to be potential limitations of the study. More comprehensive details on the uncertainties encountered in this study can be found in Coffey et al. (2015a) and (2015b). Despite these limitations, the developed modelling frameworks for all three watersheds represent useful tools that are capable of examining the impact of potential future environmental changes on the sensitivity of in-stream microbial loads.

For the existing conditions and environmental change scenarios, the Pigg River and Black watersheds violated the calendar month geometric mean standard under US CWA guidance. This indicated that reductions from microbial sources were required to meet the standard. Sources of microbial contamination in the Fergus catchment were predominantly diffuse, and the simulated microbial load was below US CWA standards for all environmental change scenarios. For existing conditions, point sources (cattle direct deposits, straight pipes, wildlife direct deposits) had the biggest impact on in-stream microbial load for the Pigg River watershed TMDL established in 2006 (Benham et al. 2006b). Some reductions in diffuse pollution (pasture and residential) were also specified in the TMDL study. Additional reductions in loads from cattle and wildlife direct deposit sources were required to meet the geometric mean and single sample standard violation rates for all future climate and land management scenarios simulated. Allocations to meet the single sample standard violation rate focused on greater reductions in loading from cattle direct deposits (up to 7 fold for some sub watersheds in comparison to the 2006 TMDL) and the exclusion of all straight pipes. For the Black watershed, reductions were concentrated on cattle direct deposits and manure application, which are the dominant microbial sources in the watershed. Large reductions in these sources were required under the various environmental change scenarios (up to 2 fold compared to existing conditions) to meet the US CWA geometric mean standard.

The subsequent application of such reductions (post modeling and TMDL establishment) typically requires the implementation of Best Management Practices (BMPs) (Copeland 2003). Extensive reductions from source loadings suggested in this study for future environmental change scenarios concurs with existing studies (Johnson and Weaver 2009), and highlights the difficulties that watershed stakeholders in the US could be tasked to manage and comply with the CWA. This could potentially necessitate expanded and more effective BMP implementation efforts to restore and maintain desired ambient water quality conditions. A summary of typical watershed BMP control measures and their microbial reduction effectiveness for diffuse and point sources of microbial contamination (Swann 1999; USEPA-CBP 2003, 2008; Yagow et al. 2009) is given in Table 9. As part of the TMDL program, implementation of these measures requires extensive funding, normally running into millions of dollars (Pandey et al. 2014). In 2003, the cost to implement TMDL plans to improve stream water was estimated as \$0.9 to \$4.3 billion per year (Copeland 2003). Therefore, extended watershed BMPs that are likely to be needed to improve and restore water quality within existing CWA standards could necessitate a substantial increase in funding and active stakeholder engagement. As the US CWAs current TMDL framework largely ignores the impacts of environmental change when developing and implementing TMDLs to restore water quality, the effectiveness of previously implemented restoration plans is uncertain (Coffey et al. 2014). Blankenship (2008) reports that anticipated climate changes could negate previous remediation efforts. Therefore, control measures that consider changes in the environment need to be included to enhance the existing TMDL framework, and would assist in maintaining the objectives of legislation going forward (Blankenship 2008; Coffey et al. 2014).

Table 9 Potential watershed Best Management Practices (BMPs) and their effectiveness in reducing microbial contamination

Control measures	Bacterial reduction effectiveness (%)	Source
Agricultural control measures		
Grass riparian buffers	48 %	1
Forested riparian buffers	48 %	1
Reforestation of erodible pasture	Simulated	2
Manure storage facilities	75 %	4
Livestock exclusion fencing	100 %	3
Livestock exclusion buffers	48 %	1
Improved pasture management	92 %	5
Residential control measures		
Pet waste education program	50 %	6
Pet waste digesters	100 %	3
System repairs	100 %	3
New septic systems	100 %	3
New septic systems w/pumps	100 %	3
Alternative septic systems	100 %	3

1 - EPA-CBP sediment effectiveness, 2008. (Bacteria efficiency assumed equal to sediment efficiency)
 2 - Based on unit bacteria load from wildlife
 3 - By definition
 4 - EPA-CBP (2003)
 5 - Yagow et al. (2009)
 6 - Modified from Swann (1999)

In contrast (to using the US CWA standards as the water quality end points), the same model outputs achieved the EU BWD classification of “excellent” during designated bathing periods (June through September) in the Fergus, Black and Pigg watersheds. This suggests that the EU BWD microbial contaminant standards are more relaxed. Given that the EU BWD only applies to freshwater designated as seasonal bathing areas, and the broad criteria metrics, it is likely that these standards are an achievable target for all freshwater under future environmental changes. However, some studies also report that insufficient attention has been given to date to the area of microbial fate and transport for watersheds in the EU (Kay et al. 2007). In addition, the potential public health risk associated with recreational water standards must also be considered when evaluating the implications of the EU BWD, US CWA and others.

The establishment of recreational water quality standards by regulatory authorities is typically based on an acceptable risk to public health. Such an approach means that there is a need for credible evidence about the risk of illness associated with increasing levels of microbial contamination under circumstantial water exposure (Kay et al. 2004). However, Kay and Fawell (2007) do suggest that historical microbiological standards are not based on robust epidemiological data. The US CWA calendar month geometric mean standard of 126 *E. coli* (CFU 100 mL⁻¹) is based on an acceptable swimming associated gastroenteritis rate of 8 per 1000 (0.8 % risk of illness) (Dufour and Ballentine 1986). Canada, like the US, also use a geometric mean and instantaneous standard in their risk management approach to safe recreational water quality. A value of 200 *E. coli* 100 mL⁻¹ is proposed as a guideline for the geometric mean standard and a value of 400 *E. coli* 100 mL⁻¹ is provided as the instantaneous standard (Tobin and Ward 1984). The guideline values for freshwater corresponds to a gastrointestinal illness rate of approximately 1–2 % (10–20 illnesses per 1000 swimmers) (Health and Canada 2012; Tobin and Ward 1984). For the EU BWD, the various water quality classifications (see Table 1) are also linked to the risk of a gastrointestinal illness. If bathing quality is deemed “excellent” for freshwater (95th% < 500 CFU 100 mL⁻¹), there is

<5 % risk of gastrointestinal illness. The status “good” (95th% < 1000 CFU 100 ml⁻¹) is related to <8 % risk of gastrointestinal illness (Ahmadi et al. 2014). This range of 95th percentiles corresponds to a probability of 50 to 80 illnesses per 1000. Therefore, the standards established in the EU BWD are more lenient in terms of the allowable in stream microbial load, however, they are also less protective of public health. The disparity in health risk underlines the more stringent nature of the US CWA when compared to other international standards. Although more protective of public health, the capacity to meet this standard for freshwater under changed environmental conditions is uncertain.

5 Conclusion

Adapting to the impacts of environmental change, in particular for water quality, is one of the most pressing challenges for water policy globally. Freshwater resources are among the systems that are particularly vulnerable. The analysis documented here represents an initial step towards understanding the possible impacts of environmental change on microbial contaminants in freshwater and the subsequent implications for recreational water use and public policy as reflected in target water quality standards. In addition, the possible scale of load reduction measures that will be required in future years to maintain microbial water quality within existing water quality standards was examined. The work focused on a variety of environmental change scenarios simulated using watershed modelling tools; thus, it should be noted that there are many uncertainties to be considered when conducting such speculative analyses. Nevertheless, the results presented here are useful and indicate increased in-stream microbial loads, due to environmental change, will lead to a decline in water quality. For US CWA guidelines, the decline could necessitate more extreme reductions from land based microbial sources to meet current ambient water quality standards. In contrast, current EU BWD standards set more easily attainable in-stream microbial concentration standards. All watersheds considered in this study achieved “excellent” status under the EU BWD for a variety of simulated environmental change scenarios. However, it should be noted that the EU BWD standards are statistically less protective of public health. Managing risks by planning for the effects of environmental change must become a critical component of watershed protection in the intermediate term. Such strategic preparation will improve existing policy making, and ensure that an optimal context is in place for future decisions based on evolving environmental conditions. This is vital to maintain the quality of all freshwater going forward, and minimize potential public health risk from pathogenic waterborne microorganisms in recreational freshwater.

Acknowledgments The authors acknowledge funding under the Marie Curie Fellowship scheme by Seventh Framework Programme of the European Union.

References

- Ahmadi M, Records R, Arabi M (2014) Impact of climate change on diffuse pollutant fluxes at the watershed scale. *Hydrol Process* 28:1962–1972. doi:10.1002/hyp.9723
- Balbus JM, Boxall ABA, Fenske RA, McKone TE, Zeise L (2013) Implications of global climate change for the assessment and management of human health risks of chemicals in the natural environment. *Environ Toxicol Chem* 32:62–78. doi:10.1002/etc.2046

- Benham BL, Baffaut C, Zeckoski RW, Mankin KR, Pachepsky YA, Sadeghi AA, Brannan KM, Soupir ML, Habersack MJ (2006a) Modeling bacteria fate and transport in watersheds to support TMDLs. *Trans ASAE* 49:987–1002
- Benham BL, Zeckoski RW, Mishra A (2006b) Bacteria total maximum daily load development for Pigg river, Snow creek, Story creek, and Old Womans creek. Virginia Tech
- Bicknell B, Imhoff J, Donigian A, Johanson R (1997) Hydrological 9 Simulation Program–FORTRAN (HSPF), User's Manual for release 11. EPA–10 600. R-97/080. United States Environmental Protection Agency, Athens, GA. 11
- Blankenship K (2008) TMDLs are coming, like it or not. <http://www.bayjournal.com/documents/reprints/TMDL.pdf>. Accessed 24 Sep 2014
- Boxall ABA, Hardy A, Beulke S, Boucard T, Burgin L, Falloon PD, Haygarth PM, Hutchinson T, Kovats RS, Leonardi G, Levy LS, Nichols G, Parsons SA, Potts L, Stone D, Topp E, Turley DB, Walsh K, Wellington EMH, Williams RJ (2009) Impacts of climate change on indirect human exposure to pathogens and chemicals from agriculture. *Environ Health Perspect* 117:508–514. doi:10.1590/S1413-81232010000300017
- Coffey R, Cummins E, Cormican M, O Flaherty V, Kelly S (2007) Microbial exposure assessment of waterborne pathogens. *Hum Ecol Risk Assess* 13:1313–1351.
- Coffey R, Cummins E, Bhreathnach N, Flaherty VO, Cormican M (2010) Development of a pathogen transport model for Irish catchments using SWAT. *Agric Water Manag* 97:101–111
- Coffey R, Dorai-Raj S, O'Flaherty V, Cormican M, Cummins E (2013) Modeling of Pathogen Indicator Organisms in a Small-Scale Agricultural Catchment Using SWAT. *Hum Ecol Risk Assess Int J* 19:232–253
- Coffey R, Benham B, Krometis L-A, Wolfe ML, Cummins E (2014) Assessing the effects of climate change on waterborne microorganisms: implications for EU and US water policy. *Hum Ecol Risk Assess Int J* 20:724–742
- Coffey R, Benham B, Kline K, Wolfe ML, Cummins E (2015a) Modeling the impacts of climate change and future land use variation on microbial transport. *Water Climate Change* 6:23. doi:10.2166/wcc.2015.049
- Coffey R, Benham B, Wolfe ML, Cummins E (2015b) Use of catchment modelling to assess the impacts of environmental change on hydrology and microbial transport in Ireland. *Regional Environmental Change Under Review*
- Collins AL, McGonigle DF (2008) Monitoring and modelling diffuse pollution from agriculture for policy support: UK and European experience. *Environ Sci Pol* 11:97–101. doi:10.1016/j.envsci.2008.01.001
- Copeland C (1999) Clean Water Act: a summary of the law. Congressional Research Service, Library of Congress, Washington, DC
- Copeland C (2003) Clean Water Act and Total Maximum Daily Loads (TMDLs) of pollutants. Congressional Research Service, Library of Congress, Washington, DC
- Crossman J, Futter MN, Oni SK, Whitehead PG, Jin L, Butterfield D, Baulch HM, Dillon PJ (2013) Impacts of climate change on hydrology and water quality: Future proofing management strategies in the Lake Simcoe watershed. *Can J Great Lakes Res* 39:19–32. doi:10.1016/j.jglr.2012.11.003
- CSO (2013) Principal CSO Agriculture Statistics. <http://www.cso.ie/en/statistics/agricultureandfishing/>. Accessed Apr 2014
- Delpia I, Jung AV, Baures E, Clement M, Thomas O (2009) Impacts of climate change on surface water quality in relation to drinking water production. *Environ Int* 35:1225–1233. doi:10.1016/j.envint.2009.07.001
- Dufour A, Ballentine R (1986) Ambient water quality criteria for bacteria, 1986: bacteriological ambient water quality criteria for marine and fresh recreational waters. National Technical Information Service, Department of Commerce, US
- Dunn SM, Brown I, Sample J, Post H (2012) Relationships between climate, water resources, land use and diffuse pollution and the significance of uncertainty in climate change. *J Hydrol* 434–435:19–35. doi:10.1016/j.jhydrol.2012.02.039
- El-Fadel M, Ghanimeh S, Maroun R, Alameddine I (2012) Climate change and temperature rise: Implications on food- and water-borne diseases. *Sci Total Environ* 437:15–21. doi:10.1016/j.scitotenv.2012.07.041
- EU (1998) COUNCIL DIRECTIVE 98/83/EC of 3 November 1998 on the quality of water intended for human consumption vol 330
- EU (2006) 7/EC of the European Parliament and of the Council of 15 February (2006) concerning the management of bathing water quality and repealing Directive 76/160/EEC vol 64
- Eurostat (2014) Agricultural production. European Commission. http://epp.eurostat.ec.europa.eu/portal/page/portal/agriculture/agricultural_production/database. Accessed May 2014
- Frey SK, Topp E, Edge T, Fall C, Gannon V, Jokinen C, Marti R, Neumann N, Ruecker N, Wilkes G, Lapen DR (2013) Using SWAT, Bacteroidales microbial source tracking markers, and fecal indicator bacteria to predict waterborne pathogen occurrence in an agricultural watershed. *Water Res* 47:6326–6337. doi:10.1016/j.watres.2013.08.010

- Gevaert V, Verdonck F, Benedetti L, De Keyser W, De Baets B (2009) Evaluating the usefulness of dynamic pollutant fate models for implementing the EU Water Framework Directive. *Chemosphere* 76:27–35. doi:10.1016/j.chemosphere.2009.02.046
- Gleeson E, McGrath R, Treanor M (2013) Ireland's climate: the road ahead. Met Éireann, Dublin
- Gray N (2014) Chapter thirty-six - the implications of global warming and climate change on waterborne diseases. In: Percival SL, Yates MV, Williams DW, Chalmers RM, Gray NF (eds) *Microbiology of Waterborne Diseases*, Second edn. Academic Press, London, pp. 653–666. doi:10.1016/B978-0-12-415846-7.00036-6
- Hampson D, Crowther J, Bateman I, Kay D, Posen P, Stapleton C, Wyer M, Fezzi C, Jones P, Tzanopoulos J (2010) Predicting microbial pollution concentrations in UK rivers in response to land use change. *Water Res* 44:4748–4759. doi:10.1016/j.watres.2010.07.062
- Hamel RD, Cooper RJ, Slade RM, Haney RL and Arnold JG (2006) Cumulative uncertainty in measured streamflow and water quality data for small watersheds. *Trans. ASAE* 49(3):689–701.
- Health, Canada (2012) Guidelines for Canadian recreational water quality, Third Edition edn. Health Canada, Ottawa
- Hofstra N (2011) Quantifying the impact of climate change on enteric waterborne pathogen concentrations in surface water. *Curr Opin Environ Sustain* 3:471–479. doi:10.1016/j.cosust.2011.10.006
- Imhoff JC, Kittle JL, Jr., Gray MR, Johnson TE (2007) Using the Climate Assessment Tool (CAT) in U.S. EPA BASINS integrated modeling system to assess watershed vulnerability to climate change. *Water Sci Technol* 56:49–56. doi: 10.2166/wst.2007.595.
- IPCC (2001) *Climate change 2001: impacts, adaptation, and vulnerability: contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge
- Johnson T, Weaver C (2009) A framework for assessing climate change impacts on water and watershed systems. *Environ Manag* 43:118–134. doi:10.1007/s00267-008-9205-4
- Johnson T, Butcher J, Parker A, Weaver C (2011) Investigating the sensitivity of U.S. streamflow and water quality to climate change: U.S. EPA Global Change Research Program's 20 Watersheds Project. *J Water Resour Plan Manag* 138:453–464. doi:10.1061/(ASCE)WR.1943-5452.0000175
- Kay D, Bartram J, Prüss A, Ashbolt N, Wyer MD, Fleisher JM, Fewtrell L, Rogers A, Rees G (2004) Derivation of numerical values for the World Health Organization guidelines for recreational waters. *Water Res* 38: 1296–1304. doi:10.1016/j.watres.2003.11.032
- Kay, D., & Fawell, J. K. (2007). Standards for Recreational Water Quality. Foundation for Water Research. <http://www.fwr.org/environw/frg0005.pdf>. Accessed July 2014
- Kay D, Edwards AC, Ferrier RC, Francis C, Kay C, Rushby L, Watkins J, McDonald AT, Wyer M, Crowther J, Wilkinson J (2007) Catchment microbial dynamics: the emergence of a research agenda. *Prog Phys Geogr* 31:59–76. doi:10.1177/0309133307073882
- Kim SM, Benham BL, Brannan KM, Zeckoski RW, Yagow GR (2007) Water quality calibration criteria for bacteria TMDL development. *Appl Eng Agric* 23:6
- Kim J-W, Pachepsky YA, Shelton DR, Coppock C (2010) Effect of streambed bacteria release on *E. coli* concentrations: monitoring and modeling with the modified SWAT. *Ecol Model* 221:1592–1604. doi:10.1016/j.ecolmodel.2010.03.005
- Kristensen P (2012) *European waters: assessment of status and pressures, 2012 edn*. Publications Office of the European Union, Luxembourg
- Kundzewicz ZW et al. (2008) The implications of projected climate change for freshwater resources and their management. *Hydrol Sci J* 53:3–10. doi:10.1623/hysj.53.1.3
- Larsen MC, Hamilton PA, Werkheiser WH (2013) 2 - Water quality status and trends in the United States. In: Ahuja S (ed) *Monitoring water quality*. Elsevier, Amsterdam, pp. 19–57. doi:10.1016/B978-0-444-59395-5.00002-9
- Macleod CJA, Falloon PD, Evans R, Haygarth PM (2012) Chapter two - the effects of climate change on the mobilization of diffuse substances from agricultural systems. In: Donald LS (ed) *Advances in agronomy*, vol 115. Academic Press, San Diego, pp. 41–77. doi:10.1016/B978-0-12-394276-0.00002-0
- Moriassi DN, Arnold JG, Van Liew MW, Bingner RL, Harmel RD, Veith TL (2007) Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Trans ASABE* 50:16
- Murdoch PS, Baron JS, Miller TL (2000) Potential effects of climate change on surface-water quality in North America. *JAWRA J Am Water Resour Assoc* 36:347–366. doi:10.1111/j.1752-1688.2000.tb04273.x
- Najjar RG, Pyke CR, Adams MB, Breitburg D, Hershner C, Kemp M, Howarth R, Mulholland MR, Paolisso M, Secor D, Sellner K, Wardrop D, Wood R (2010) Potential climate-change impacts on the Chesapeake Bay Estuarine. *Coast Shelf Sci* 86:1–20. doi:10.1016/j.ces.2009.09.026
- Oliver DM, Van Niekerk M, Kay D, Heathwaite AL, Porter J, Fleming LE, Kinzelman JL, Connolly E, Cummins A, McPhail C, Rahman A, Thairs T, de Roda Husman AM, Hanley ND, Dunhill I, Globevnik L, Harwood VJ, Hodgson CJ, Lees DV, Nichols GL, Nocker A, Schets C, Quilliam RS (2014) Opportunities and

- limitations of molecular methods for quantifying microbial compliance parameters in EU bathing waters. *Environ Int* 64:124–128. doi:10.1016/j.envint.2013.12.016
- Palmer MA, Reidy Liermann CA, Nilsson C, Flörke M, Alcamo J, Lake PS, Bond N (2008) Climate change and the world's river basins: anticipating management options. *Front Ecol Environ* 6:81–89. doi:10.1890/060148
- Pandey PK, Soupir ML, Rehmann CR (2012) A model for predicting resuspension of *Escherichia coli* from streambed sediments. *Water Res* 46:115–126. doi:10.1016/j.watres.2011.10.019
- Pandey P, Kass P, Soupir M, Biswas S, Singh V (2014) Contamination of water resources by pathogenic bacteria. *AMB Express* 4:1–16. doi:10.1186/s13568-014-0051-x
- Patz JA, Campbell-Lendrum D, Holloway T, Foley JA (2005) Impact of regional climate change on human health. *Nature* 438:310–317
- Piorowski G, Jamieson R, Bezanson G, Truelstrup Hansen L, Yost C (2014) Reach specificity in sediment *E. coli* population turnover and interaction with waterborne populations. *Sci Total Environ* 496:402–413. doi:10.1016/j.scitotenv.2014.06.145
- Quevauviller P (2011) Adapting to climate change: reducing water-related risks in Europe – EU policy and research considerations. *Environ Sci Pol* 14:722–729. doi:10.1016/j.envsci.2011.02.008
- Radcliffe DE, Freer J, Schoumans O (2009) Diffuse phosphorus models in the United States and Europe: their usages, scales, and uncertainties. *J Environ Qual* 38:1956–1967. doi:10.2134/jeq2008.0060
- Russo S, Hunn J, Charakkis G (2011) Considering bacteria-sediment associations in microbial fate and transport modeling. *J Environ Eng* 137:697–706. doi:10.1061/(ASCE)EE.1943-7870.0000363
- Semenza JC, Menne B (2009) Climate change and infectious diseases in Europe. *Lancet Infect Dis* 9:365–375. doi:10.1016/S1473-3099(09)70104-5
- Soller JA, Schoen ME, Bartrand T, Ravenscroft JE, Ashbolt NJ (2010) Estimated human health risks from exposure to recreational waters impacted by human and non-human sources of faecal contamination. *Water Res* 44:4674–4691. doi:10.1016/j.watres.2010.06.049
- St. Laurent J, Mazumder A (2014) Influence of seasonal and inter-annual hydro-meteorological variability on surface water fecal coliform concentration under varying land-use composition. *Water Res* 48:170–178. doi:10.1016/j.watres.2013.09.031
- Swann C (1999) A survey of residential nutrient behaviors in the Chesapeake Bay. Widener Burrows, Inc. Chesapeake Research Consortium Center for Watershed Protection, Ellicott City
- Tobin RS, Ward W (1984) Guidelines for Canadian recreational water quality. *Can J Public Health* 75(1):15–18
- USEPA (2004) National water quality inventory. United States Environmental Protection Agency, Office of Water, Washington, DC 20460
- US EPA (2012) Recreational Water Quality Criteria. United States Environmental Protection Agency, Office of Water, Washington, DC 20460
- USEPA-CBP (2003) Chesapeake Bay Program Phase 4.3 Watershed Model nonpoint source BMPs
- USEPA-CBP (2008) Agricultural BMP effectiveness estimates
- Vermeulen L, Hofstra N (2014) Influence of climate variables on the concentration of *Escherichia coli* in the Rhine, Meuse, and Drentse Aa during 1985–2010. *Reg Environ Chang* 14:307–319. doi:10.1007/s10113-013-0492-9
- Wilby RL, Orr HG, Hedger M, Forrow D, Blackmore M (2006a) Risks posed by climate change to the delivery of Water Framework Directive objectives in the UK. *Environ Int* 32:1043–1055. doi:10.1016/j.envint.2006.06.017
- Wilby RL, Whitehead PG, Wade AJ, Butterfield D, Davis RJ, Watts G (2006b) Integrated modelling of climate change impacts on water resources and quality in a lowland catchment: River Kennet, UK. *J Hydrol* 330:204–220. doi:10.1016/j.jhydrol.2006.04.033
- Winchell M, Srinivasan R, Di Luzio M, Arnold J (2007) Arc-SWAT interface for SWAT2005-User's guide. USDA Agricultural Research Service and Texas A&M Blackland Research Center, Temple
- Yagow G, Benham B, Sieber T, Mizel M, Faye C, Dukes F, Wilkinson C (2009) Smith Creek watershed TMDL implementation plan