Science of the Total Environment 426 (2012) 32–44

Contents lists available at SciVerse ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

The effect of risk perception on public preferences and willingness to pay for reductions in the health risks posed by toxic cyanobacterial blooms

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ARTICLE INFO

Article history:
Received 11 October 2011
Received in revised form 19 January 2012
Accepted 9 February 2012
Available online 20 April 2012

Keywords:
Environmental economics
Water quality
Recreational waters
Spike model
Endogeneity bias
Catchment management

ABSTRACT

Mass populations of toxin-producing cyanobacteria are an increasingly common occurrence in inland and coastal waters used for recreational purposes. These mass populations pose serious risks to human and animal health and impose potentially significant economic costs on society. In this study, we used contingent valuation (CV) methods to elicit public willingness to pay (WTP) for reductions in the morbidity risks posed by blooms of toxin-producing cyanobacteria in Loch Leven, Scotland. We found that 55% of respondents (68% excluding protest voters) were willing to pay for a reduction in the number of days per year (from 90, to either 45 or 0 days) that cyanobacteria pose a risk to human health at Loch Leven. The mean WTP for a risk reduction was UK£9.99–12.23/household/year estimated using a logistic spike model. In addition, using the spike model and a simultaneous equations model to control for endogeneity bias, we found the respondents' WTP was strongly dependent on socio-demographic characteristics, economic status and usage of the waterbody, but also individual-specific attitudes and perceptions towards health risks. This study demonstrates that anticipated health risk reductions are an important nonmarket benefit of improving water quality in recreational waters and should be accounted for in future cost–benefit analyses such as those being undertaken under the auspices of the European Union's Water Framework Directive, but also that such values depend on subjective perceptions of water-related health risks and general attitudes towards the environment.

1. Introduction

1.1. Cyanobacteria and human health

Cyanobacteria (blue-green algae) are a diverse group of naturally occurring, Gram-negative photosynthetic microorganisms found almost ubiquitously in fresh, transitional and marine waters. They form an integral component of the microbial biodiversity of aquatic ecosystems and also fulfil key functional roles in biogeochemical cycling (Whitton and Potts, 2000). However, in warm and nutrient-enriched waters, cyanobacteria readily form mass populations as planktonic blooms, surface scums and benthic mats. These mass populations can pose serious risks to human and animal health because cyanobacteria are capable of producing a range of bioactive toxins (cyanotoxins) that have been shown to have neurotoxic, hepatotoxic and tumour-promoting, cytotoxic, genotoxic and endotoxic properties. The toxicity, speed and mode of action of these toxins vary greatly but they include some of the most potent of all bioproducts found in inland waterbodies (Codd et al., 2005).

Mass populations of toxin-producing cyanobacteria are increasing globally in inland and coastal waters because of nutrient inputs from human activities and sources (e.g. agriculture, industry, sewage) (Smith, 2003) and recent climate warming (Paerl and Huisman, 2008). Affected waters include those used for drinking water supplies, livestock watering, fishing, crop irrigation and recreation (Codd et al., 2005). Human exposure to cyanobacterial blooms and their toxins is potentially widespread and can occur through several routes including accidental and/or incidental dermal contact, ingestion and inhalation during recreational and occupational activities (Pilotto et al., 2004; Stewart et al., 2006; Caller et al., 2009) and through the consumption of ineffectively-treated drinking water (Falconer et al., 1983), shellfish and finfish (Falconer et al., 1992) and spray-irrigated crops (Codd et al., 1999; Crush et al., 2008; Tyler et al., 2009).

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0048-9697/$ – see front matter
doi:10.1016/j.scitotenv.2012.02.017

There is an increasing body of evidence ascribing a range of adverse human health outcomes to acute and chronic exposure to cyanotoxins. These include dermatological (Pilotto et al., 2004; Stewart et al., 2006), respiratory (Turner et al., 1990) and gastro-intestinal effects (Teixeira et al., 1993) as well as acute liver failure (Carmichael et al., 2001). While the most frequently reported ill health effects comprise relatively minor hay fever-like symptoms, pruritic skin rashes and gastroenteritis, several case reports also document incidences of acute (Turner et al., 1990) and lethal toxicoses (Carmichael et al., 2001; Stewart et al., 2006). Research has also linked cyanotoxins to clusters of primary liver cancer (Svircev et al., 2009) and as possible potentiators of neurodegenerative disease (Amyotrophic Lateral Sclerosis: Motor Neurone Disease) (Ince and Codd, 2005; Metcalf and Codd, 2009).

The long-term control of cyanobacterial populations and their associated health risks can only be achieved effectively through the reduction of internal and external nutrient loads (primarily P and N) to waterbodies (Codd et al., 2005). Such reductions are costly, whether in terms of investments in improved sewage treatment, in costs of changes in catchment nutrient management (for example, by farmers), or investments in waterbody remediation. In Europe, the Water Framework Directive (WFD) (2000/60/EC) is the flagship policy intended to deliver improvements in the status of inland surface waters. The central aim of the WFD is achieve “good ecological status” in all surface (and ground) waters by 2015. The WFD stands apart as the first European Directive to explicitly recognise the role of economics in achieving environmental quality objectives through its requirement that Member States assess the economic costs and benefits of measures of achieving good ecological status during the formulation of River Basin Management Plans (RBMPs).

It is widely recognised that the declining status of surface waters globally imposes substantial economic costs on society (Pretty et al., 2003; Dodds et al., 2009) and thus the anticipated improvements in the ecological status of Europe's surface waters under the WFD can be expected to generate significant economic benefits. It has been widely shown that nonmarket values are likely to represent a significant component of these benefits (Hanley et al., 2006b; Del Saz-Salazar et al., 2009; Martin-Ortega and Berbel, 2010). The mitigation of water-related human health risks is one such nonmarket benefit that might arise through efforts to improve the ecological status of Europe's surface waters. However, estimates of the welfare benefits of water-related health risk mitigations are currently incomplete. In particular, few studies have attempted to estimate the benefits of reducing health risks posed by toxigenic cyanobacteria in waterbodies used for drinking and/or recreation. Such benefit estimates are, however, necessary, if cost–benefit comparisons of water quality improvements are to be undertaken. This paper presents such benefit estimates for a high-resource waterbody in Scotland impacted by blooms of toxin-producing cyanobacteria. Additionally, we investigate the effect of changes in risk perception on the local population's willingness to pay (WTP) for measures to reduce the occurrence of cyanobacteria blooms of human health significance.

1.2. Estimating the benefits of reductions in cyanobacteria

The economic value of changes in ecosystem services can be estimated through a range of methods (Hanley and Barbier, 2009). These include: stated preference methods such as CV and choice experiments; revealed preference methods such as hedonic pricing and travel cost models; and production function approaches. Choosing an appropriate method for a given empirical setting requires a consideration of the nature of environmental benefits or costs to be measured (UK NEA, 2011). Where variations in environmental quality solely impact on users of a particular environmental resource, then revealed preference methods are often a preferred approach. However, when benefits are likely to accrue to both users and non-users of the resource, or where changes in environmental quality beyond the range of current variation are in prospect, a stated preference method is more appropriate (Hanley and Barbier, 2009). Previous studies of the economic benefits of water quality improvements have shown both use and non-use values to be important (Hanley et al., 2003a; Holmes et al., 2004; Birol et al., 2006). Given our expectation that both those who directly use our case study waterbody (e.g. for fishing) and those who do not directly use the waterbody might have positive values for health risk reductions, CV – a stated preference method – was chosen for the present study.

The maxim of CV is that an individual, when presented with a contingent (i.e. hypothetical) market for a specific change in an environmental good or service, will reveal their underlying preference for that change through their survey response (Bateman et al., 2002). The value that an individual places on a nonmarket environmental good can be estimated from the largest amount of money they would be prepared to pay for the delivery of the environmental good, their maximum WTP (or alternatively the lowest amount they would be prepared to accept to forgo that same good). In essence, an individual's WTP reflects the trade-off between the consumption of other goods and the proposed environmental benefit. Aggregating WTP amounts across the beneficiaries of a change in environmental quality yields an estimate of the total economic benefit of that change.

Formally, if an individual, *i*'s, utility is derived from the consumption of a vector of privately supplied goods available at market prices *P*, a vector of public (environmental) goods *Q*, and available income *m*, we can write the (indirect) utility function as:

$$v(P, Q, m). \tag{1}$$

If the quantity of an environmental good such as safe water increases from Q_0 to Q_1, then the associated welfare gain (consumer surplus or WTP) can be quantified as:

$$v(P_0, Q_0, m_0) = v(P_0, Q_1, m_0 - \mathrm{WTP}). \tag{2}$$

WTP represents the reduction in income available to spend on market goods necessary to offset the increase of water quality from Q_0 to Q_1.

In a random utility model, individual *i*'s indirect utility function can be specified as combining an observable, deterministic component with a random component (*e*), which is un-observable (McFadden, 1974):

$$v_i(P_i, Q_i, m_i) = V_i(Z_i, Q_i, m_i, \varepsilon_i) \tag{3}$$

We can now omit the price vector (as market prices remain constant in the hypothetical choices respondents make) and include a vector of individual-specific attributes, *Z*, which includes socio-demographic characteristics of the respondent. The new random component of the utility function, ε, reflects the researcher's inability to observe all characteristics influencing the individual's utility function, and hence fully deterministically predict choices, which remain to some extent random.[1]

As noted above, stated preference methods such as CV have been widely used to determine public preferences and WTP for improvements in the quality of surface waters. This includes studies on the nonmarket benefits of improving the status of rivers and lakes (Cooper et al., 2004; Hanley et al., 2006b) and coastal bathing waters (Georgiou et al., 1998; Hanley et al., 2003a). Van Houtven et al. (2007) report a meta-analysis of studies investigating public preferences for water quality improvements in the US and found WTP

[1] Otherwise the model could not deal with different choices made by two seemingly identical (in terms of observed characteristics) individuals.