An Assessment of Potential Exposure and Risk from Estrogens in Drinking Water

Caldwell, Daniel J.

National Institute of Environmental Health Sciences


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Boston University
BACKGROUND: Detection of estrogens in the environment has raised concerns in recent years because of their potential to affect both wildlife and humans. The incomplete removal by publicly owned treatment works (POTWs) of excreted endogenous estrogens and prescribed estrogens leads to their introduction into surface waters and potentially into drinking water sources that rely on surface water. Estrogens, specifically estrone (E1), 17β-estradiol (E2), estriol (E3), and ethinyl estradiol (EE2), have been detected in numerous studies of wastewater influents and effluents (Baronti et al. 2000; Belfroid et al. 1999; Desbrow et al. 1998; Ferguson et al. 2001; Heberer 2002; Huang and Sedlak 2001; Huggett et al. 2003; Lagana et al. 2000; Moutatissim-Souali et al. 2003; Nasu et al. 2001; Rodgers-Gray et al. 2000; Spengler et al. 2001; Ternes et al. 1999a, 1999b, 2002), and their presence has been confirmed in U.S. and European surface waters (Aherne and Briggs 1989; Belfroid et al. 1999; Heberer 2002; Kolodziej et al. 2003; Kolpin et al. 2002; Kuch and Ballschmiter 2001).

More recently, several estrogens have also been detected in the source water of drinking water treatment plants but not in the finished water (Benotti et al. 2009).

The effects of estrogens on fish and other aquatic organisms have been widely studied (see Caldwell et al. 2008). However, fewer studies have evaluated the potential effects of estrogens in surface water on humans. Moreover, the available studies on exogenous estrogens reach differing conclusions on potential human effects (Aherne and Briggs 1989; Andersson and Skakkebaek 1999; Christensen 1998; Webb et al. 2003). Based on independent worst-case exposure estimates, Aherne and Briggs (1989) and Christensen (1998) concluded that risks from environmental sources of the synthetic hormone EE2 were negligible compared with normal body concentrations of estrogens. Webb et al. (2003) noted that worst-case indirect exposure to EE2 via drinking water would be three to four orders of magnitude lower than endogenous production rates of E2. Andersson and Skakkebaek (1999), however, raised concern that there may be no threshold level regarding the action of exogenous estrogens, particularly for prepubertal males. Previous reports (Fritsche and Steinhart 1999; Hartmann et al. 1998) concluded that dietary intake of the endogenous estrogens is minimal compared with human production rates.

For the most part, articles reporting detection of estrogens in surface and drinking waters both in the public press (Donn et al. 2008a, 2008b, 2008c) and in the scientific literature (reviewed by Ying et al. 2002), provide little comparison as to whether potential drinking water exposures are large or small compared with other sources of exposure (e.g., dietary intake) or compared with acceptable daily intakes (ADIs). This makes it difficult to determine whether exposures from drinking water derived from surface water are a significant source of overall estrogen exposure or have the potential to exceed ADIs and, thus, whether they deserve additional evaluation.

The analysis presented here is, to our knowledge, the first exposure assessment for estrogens in drinking water that distinguishes among the potential sources of estrogens. In an earlier human health risk assessment of pharmaceuticals in U.S. surface waters, Schwab

### An Assessment of Potential Exposure and Risk from Estrogens in Drinking Water

Daniel J. Caldwell,1 Frank Mastrocco,2 Edward Nowak,3 James Johnston,4 Harry Yekel,2 Danielle Pfeiffer,5 Marilyn Hoyt,6 Beth M. DuPlessie,6 and Paul D. Anderson6,7

1Johnson & Johnson Worldwide Environment, Health, and Safety, New Brunswick, New Jersey, USA; 2Pfizer Inc., New York, New York, USA; 3Johnson & Johnson Pharmaceutical Research and Development, Raritan, New Jersey, USA; 4Wyeth, Madison, New Jersey; 5ARCADIS, Chelmsford, Massachusetts, USA; 6AMEC Earth & Environmental, Westford, Massachusetts, USA; 7Department of Geography and Environment, Boston University, Boston, Massachusetts, USA

Detection of estrogens in the environment has raised concerns in recent years because of the potential of these compounds to affect both wildlife and humans. The incomplete removal by publicly owned treatment works (POTWs) of excreted endogenous estrogens and prescribed estrogens leads to their introduction into surface waters and potentially into drinking water sources that rely on surface water. Estrogens, specifically estrone (E1), 17β-estradiol (E2), estriol (E3), and ethinyl estradiol (EE2), have been detected in numerous studies of wastewater influents and effluents (Baronti et al. 2000; Belfroid et al. 1999; Desbrow et al. 1998; Ferguson et al. 2001; Heberer 2002; Huang and Sedlak 2001; Huggett et al. 2003; Lagana et al. 2000; Moutatissim-Souali et al. 2003; Nasu et al. 2001; Rodgers-Gray et al. 2000; Spengler et al. 2001; Ternes et al. 1999a, 1999b, 2002), and their presence has been confirmed in U.S. and European surface waters (Aherne and Briggs 1989; Belfroid et al. 1999; Heberer 2002; Kolodziej et al. 2003; Kolpin et al. 2002; Kuch and Ballschmiter 2001).

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Address correspondence to P.D. Anderson, ARCADIS U.S. Inc., 2 Executive Dr., Suite 303, Chelmsford, MA 01824 USA. Telephone: (978) 937-9999, ext. 304. Fax (978) 937-7555. E-mail: paul.anderson@arcadis-us.com

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et al. (2005) found no appreciable health risk from 26 drugs representing 14 different drug classes, but estrogens were not included in the study. In the present study we used a weight-of-evidence approach to determine whether predicted exposure to trace levels of prescribed and naturally occurring estrogens in drinking water has the potential to cause effects. We developed several lines of evidence which fall into two general categories.

The first category consists of comparing a typical U.S. resident’s potential drinking water exposures with background dietary exposures (e.g., exposure via consumption of milk or in the overall diet). Naturally occurring, animal-derived estrogens (e.g., E1, E2, and E3) have been measured in a wide variety of foods that are regularly consumed by most Americans (Doyle 2000; Fritsche and Steinhart 1999; Hartmann et al. 1998; Henricks et al. 1983; Tsujioka et al. 1992). Dietary intake is likely the dominant pathway of estrogen exposure for most people in the United States of both sexes and all ages (except women using prescribed estrogens for birth control, hormone therapy, or hormone replacement therapy) (Fritsche and Steinhart 1999). Although we do not know whether dietary exposures are or are not associated with effects (adverse or beneficial), we do know they represent a consistent daily exposure for U.S. residents. Whether drinking water exposures are large or small compared with dietary exposure to these same estrogens provides important perspective as to their relative significance.

The second category of lines of evidence consists of comparing drinking water exposures with toxicity-based benchmarks assumed to be without adverse effect [e.g., the World Health Organization (WHO) ADI, the threshold for toxicologic concern (TTC), occupational exposure limits, and Australian Guidelines for Water Recycling (Environment Protection and Heritage Council (EPHC) et al. 2008)]. These benchmarks represent estimates of exposure that are assumed to be safe and are derived using commonly accepted public health practices. Whether drinking water exposures are larger or smaller than these benchmarks indicates whether they may or may not be associated with adverse effects.

Methods

Estimating exposure to estrogens via drinking water requires information about the concentrations of estrogens in drinking water and the amount of water consumed by a typical person in the United States. The U.S. Environmental Protection Agency (EPA) recommended water ingestion rates of 0.87 L/day for children and 1.4 L/day for adults were used to estimate drinking water consumption (U.S. EPA 1997). Predicted concentrations in drinking water were used instead of measured concentrations because few studies have measured estrogen concentrations in U.S. drinking water, and those that are available report primarily nondetected concentrations [see Supplemental Material, available online (doi:10.1289/ehp.0900654.S1) via http://dx.doi.org/]; see also Hannah et al. 2009]. The predicted environmental concentrations (PECs) of synthetic estrogens and endogenous estrogens in drinking water resulting from human use and excretion were estimated using the PdATE (Pharmaceutical Assessment and Transport Evaluation) model, version 2.1.1 (Anderson et al. 2004). PdATE requires several compound-specific inputs, including the per capita use or excretion rate, metabolism, POTW removal rate, in-stream removal, and drinking water treatment removal. Removal rates were not available for all estrogens. When no removal information was available for a particular estrogen and a particular removal mechanism, we assumed that no removal occurred to assure that drinking water PECs were not underestimated. A summary of PdATE inputs is provided in the Supplemental Material, Tables SM-4 through SM-9 (doi:10.1289/ehp.0900654.S1).

PdATE generates PECs for approximately 2,710 stream segments in 11 U.S. watersheds and for 184 drinking water treatment plants serving approximately 10,800,000 people located on the modeled stream segments. The PECs used in this analysis are those for each of the drinking water treatment systems included in PdATE. PdATE is able to generate PECs for both mean flow conditions and 7Q10 low-flow (i.e., the lowest consecutive 7-day low flow that occurs on average once every 10 years) conditions. To be conservative, in this analysis we used 7Q10 low-flow PECs because this is when estrogen concentrations in surface water used as source water for a drinking water treatment plant would be at their highest (as opposed to annual average flow when concentrations would be lower).

PECs generated by PdATE are grouped five ways to enable discrimination among different sources of estrogens in drinking water and among types of estrogens. Potential exposures are presented for prescribed synthetic estrogen alone (i.e., EE2), prescribed endogenous estrogens (i.e., E1, E2, and E3 prescribed for therapeutic use), total prescribed estrogens (i.e., E1, E2, E3, and EE2 prescribed for therapeutic use), naturally occurring endogenous estrogens (i.e., naturally occurring animal-derived E1, E2, and E3), and total estrogens (E1, E2, E3, and EE2 from all sources). For dietary and ADI comparisons that required combining exposure to individual estrogens, total estrogen exposure was expressed as estradiol equivalents (E2-eq), which were estimated based on alpha receptor binding for E1, E2, and E3 (assumed to have relative biological activity of 0.1, 1.0, and 0.038, respectively). To avoid potentially underestimating the estrogenic activity of synthetic estrogen, EE2 was assumed to have 10 times the activity of E2 (i.e., a relative potency of 10). More detailed discussion of the relative biological activity adjustments is presented in Supplemental Material (doi:10.1289/ehp.0900654.S1).

Dietary benchmarks. Estrogens have specific direct and indirect effects that may be functions of age as well as sex (Aksglaede et al. 2006; Andersson and Skakkebaek 1999; Dey et al. 2000). Some potentially sensitive subpopulations include prepubescent males who have low natural serum estrogen levels and an expected low metabolic clearance rate (Andersson and Skakkebaek 1999; Klein et al. 1996), postmenopausal women with naturally low estrogen levels, and people with specific dietary preferences, such as vegetarians, that increase their exposure to phytoestrogens. We examined the differences in total intake of estrogens among several different age groups, between males and females, and among dietary preferences. Women taking prescribed estrogens for therapeutic use had the largest total daily exposure to estrogens, followed by infants fed soy-based formula, then by infants on breast milk or milk-based formula, and last by children and adults (data not shown).

Given the concern about exposure of prepubescent males (an assumed sensitive subpopulation) to estrogens in drinking water, this analysis used estrogen exposure from milk consumption of young children as one set of dietary benchmarks. The second set of dietary benchmarks is the daily exposure of an adult female eating an omnivorous diet. This is likely to be representative of the daily estrogen exposure for the majority of adults in the United States.

We estimated dietary exposure to endogenous estrogens by combining the reported concentrations of estrogens in foodstuffs with the average consumption rate of the foodstuff [see Supplemental Material, Tables SM-1 through SM-3 (doi:10.1289/ehp.0900654.S1)]. Endogenous estrogen intake is likely biased low because concentration data for one or more estrogens in a particular food may be absent and data for other foodstuffs expected to contain estrogens have not been published. Additionally, this analysis does not include the contribution of phytoestrogens to the background estrogenic activity present in a typical U.S. diet. Given that adult premenopausal women are reported to have a total isoflavone intake of 1.78 mg/day (Horn-Ross et al. 2002) and 2.17 mg/day (Huang et al. 2000) and a total lignan intake of 0.108 mg/day (Horn-Ross et al. 2002), 0.525 mg/day (McCann et al. 2003), and 0.645 mg/day (deKleijn et al. 2002), omission of phytoestrogens understates background dietary estrogen exposure. Although biologically active (Masutomi et al. 2004; Montani et al. 2008; Rimoldi et al. 2007),
phytoestrogens were excluded because data on their concentrations in many foods are not available and their estrogenic potency relative to the animal-derived endogenous estrogens remains difficult to quantify.

**Toxicity-based benchmarks.** In addition to comparing estrogen exposure via drinking water with background dietary exposure, this analysis compares exposure to estrogens in drinking water with four independently derived ADIs (or sets of ADIs) available in the literature.

The WHO (2000) derived an ADI of 0.05 µg E2/kg body weight (BW)/day based on a no observed effect level (NOEL) of 0.3 mg E2/person/day associated with changes in several hormone-dependent parameters. The WHO divided the NOEL by an uncertainty factor (UF) of 10 to account for “normal variation” among individuals and a second UF of 10 to account for “sensitive populations.” In the United States, the assumed adult body weight is 60 kg, so the adjusted WHO whole-body ADI is 3 µg E2/person/day.

The TTC is an acceptable daily exposure presented by Kroes et al. (2004) based on a review of toxicity data from a variety of chemicals. In the present study, we conservatively assumed that estrogens are structurally active compounds and evaluated them using the TTC of 0.15 µg/person/day derived by Kroes et al. (2004) for compounds with a structural alert. Such compounds were assigned the lowest TTC. Had a higher (i.e., less conservative) TTC been used, the MOS estimated in this analysis would have been higher.

Because EE2 is produced in a manufacturing environment, its makers developed occupational exposure limits for the protection of workers using all available toxicologic and pharmacologic data to protect against potential hazards, primarily from dust inhalation. The myriad biologic responses to estrogen exposure argue for use of such an integrated assessment of effect in risk assessment. The occupational exposure limit is established to protect humans against all biologically significant effects and is based on *in vivo* studies and human experience. The EE2 occupational exposure limit of 0.01 µg/m³ used in our analysis is the most recent and lowest of five occupational exposure limits developed by different manufacturers (Johnson & Johnson, unpublished data). It was converted to an allowable dose of 0.07 µg EE2/person/day by adjusting from an allowable air concentration to an ADI by multiplying by an assumed inhalation rate of 10 m³/person/day and multiplying by 5/7 to account for the difference in number of days a worker is exposed per week versus a member of the general public. An additional 10-fold reduction to account for sensitive populations, in this case potential effects on the developing infant, results in an ADI of 0.007 µg EE2/person/day. Similarly, ADIs of 0.07, 0.02, and 0.07 µg/person/day for E1, E2, and E3, respectively, were derived from their respective occupational exposure limits (0.1 µg/m³, 0.029 µg/m³, and 0.1 µg/m³ (Caldwell DJ, personal communication; Johnson & Johnson, unpublished data)) using the same approach as presented for EE2.

Australia developed water reuse guidelines for estrogens (EPHC et al. 2008). Australia used the WHO ADI to develop the E2 guideline; however, the ADIs for E1, E3, and EE2 were derived by applying a 10,000-fold safety factor to the lowest therapeutic dose, including a safety factor of 10 to account for sensitive populations. The resulting ADIs for E1, E2, E3, and EE2 are 0.052, 3, 0.084, and 0.0026 µg/person/day, respectively.

**Results**

**PECs generated by PhA TE.** PhA TE’s ability to predict representative surface water concentrations has been documented previously for a variety of compounds (Anderson et al. 2004) and more recently for EE2 in a critical review comparing surface water PECs with all available measured concentrations of EE2 in surface water (Hannah et al. 2009). Drinking water PECs are slightly lower than surface water PECs because drinking water intakes are present on < 10% of stream segments, and these segments are unlikely to be immediately downstream of POTWs. Segments immediately downstream of POTWs have the highest surface water PECs.

The PhA TE model is able to generate PECs associated with various sources of estrogens to drinking water. The ability to distinguish the relative contribution of different sources points out a unique benefit of modeling concentrations because it is not possible through measurement, for example, to distinguish prescribed E2 from naturally occurring E2 in a water sample. Excreted naturally occurring endogenous estrogens have the highest drinking water PECs, followed by prescribed endogenous estrogens. Prescribed synthetic estrogens (i.e., EE2) have the lowest PECs (Table 1, Figure 1).

We estimated drinking water exposures using the arithmetic mean of drinking water PECs assuming 7Q10 low-flow conditions (Table 1, Figure 1). The arithmetic mean low-flow PEC represents the 79th, 78th, and 80th percentile of the cumulative drinking water system PECs for naturally occurring endogenous, prescribed endogenous, and prescribed synthetic estrogens, respectively. Use of the

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**Table 1. Summary of PECs for three categories of estrogens in U.S. drinking water.**

<table>
<thead>
<tr>
<th>Category, compound</th>
<th>90th percentile</th>
<th>Average</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mean flow</td>
<td>Low flow</td>
</tr>
<tr>
<td>Endogenous estrogens from diet and naturally produced</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E1</td>
<td>0.01</td>
<td>0.1</td>
</tr>
<tr>
<td>E2</td>
<td>0.02</td>
<td>0.19</td>
</tr>
<tr>
<td>E3</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>Prescribed endogenous estrogens</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E1</td>
<td>0.02</td>
<td>0.18</td>
</tr>
<tr>
<td>E2</td>
<td>0.000015</td>
<td>0.000013</td>
</tr>
<tr>
<td>Prescribed synthetic estrogens</td>
<td></td>
<td></td>
</tr>
<tr>
<td>EE2</td>
<td>0.0003</td>
<td>0.005</td>
</tr>
</tbody>
</table>

**Figure 1.** Cumulative distribution (and arithmetic mean) of PECs generated by PhA TE for three different categories of estrogens in U.S. drinking water assuming critical low-flow conditions (7Q10). For the endogenous estrogens, the combined concentrations of E1, E2, and E3 were not adjusted for differences in biological activity.
arithmetic mean low-flow PEC leads to conservative but not extreme estimates of potential drinking water exposure and is consistent with the use of mean, rather than upper bound or maximum, concentrations of endogenous estrogens in foodstuffs. Most of the time, concentrations in drinking water will be lower because actual flow will be higher.

Comparison of drinking water to dietary exposures. We present two sets of dietary comparisons. To address concerns about the potential exposures of preadolescent children to estrogens predicted to be in drinking water, in the first set we compared a young child’s exposure to estrogens in drinking water with her or his dietary exposure to naturally occurring estrogens in milk (milk consumption is encouraged in young children) (Department of Health and Human Services 2005). To evaluate exposure of the general population, we compared an adult’s predicted exposure to estrogens via drinking water with an omnivore’s exposure to naturally occurring estrogens in the overall diet.

The results of the comparison of drinking water to dietary estrogen exposures are referred to as margins of exposure (MOEs). For the young child, we present MOEs for individual estrogens as well as all estrogens combined. Presenting MOEs on an individual estrogen basis allows for the derivation of MOEs without using relative potency adjustments for the endogenous estrogens, thus eliminating the uncertainty associated with such adjustment factors. A young child’s exposure to naturally occurring E1, E2, and E3 through typical consumption of milk [assumed to be about 0.42 L/day; see Supplemental Material (doi:10.1289/ehp.0900654.S1)] is approximately 730, 5,000, and 480,000 times, respectively, greater than his or her exposure to trace concentrations of those estrogens predicted to be in drinking water as a result of human therapeutic use (Figure 2). The MOEs for naturally occurring endogenous estrogens in drinking water (ranging from approximately 100 to 600; Figure 2) are smaller than the MOEs for prescribed estrogens, indicating that natural sources of E1, E2, and E3 contribute more to drinking water exposures than do prescribed sources.

Because EE2 does not occur naturally in milk, we compared a child’s predicted exposure to EE2 via drinking water (expressed as E2-eq) with the E2-eq concentration in milk. The E2 drinking water MOE is about 250 (i.e., a child’s exposure to EE2 in drinking water is about 250-fold smaller than his or her E2-eq exposure from drinking milk; Figure 2). Even when all sources of estrogens in drinking water are considered, the E2-eq exposure from drinking water for a young child is about 150 times lower than the exposure from milk alone (Table 2).

Comparison of an adult’s potential exposure to estrogens via drinking water with overall dietary intake of E1, E2, and E3 reveals that the MOEs for predicted drinking water intake vary from 82 to 1,700 depending on estrogen category (Table 2). Prescribed endogenous estrogens have the largest MOE (1,700), followed by naturally occurring endogenous estrogens (MOE = 220), prescribed synthetic estrogens (MOE = 140), total prescribed (MOE = 130), and finally, total estrogens (MOE = 82; Table 2).

Comparison of drinking water exposures to toxicity-based benchmarks. The results of the comparisons of drinking water estrogen intake to toxicity-based benchmarks are referred to as margins of safety (MOSs), which provide an estimate of how many times smaller the predicted drinking water intake is than the toxicity-based benchmark. When an adult’s potential prescribed estrogen exposures (expressed as E2-eq) are compared with the WHO ADI of 3 µg E2/person/day, the MOS for total prescribed estrogens, respectively, are about 840, 440,000, 3,700,000, and 160, respectively, and the combined MOS is 135 (Figure 3). MOSs for young children are approximately two times smaller than those for adults because young children are assumed to consume about two times more water on a per kilogram basis than do adults. The lowest MOSs for children result from comparing total prescribed estrogens (MOS = 55) and total of all estrogens combined (prescribed and naturally occurring, MOS = 28) in drinking water with ADIs derived from the Australian guidelines.

The WHO ADI is derived by applying commonly used UFs to therapeutic doses given to postmenopausal women. Although one of these UFs was intended to account for sensitive individuals, it is possible that a UF of 10 may not fully account for differences in sensitivity between postmenopausal women and either men/boys or children? Recent multigenerational studies have examined the effects of exposure to either E2 or EE2 during gestation and early life stages on the reproductive system of young male rodents (Howdeshell et al. 2008; Latendresse et al. 2009; Tyl et al. 2008). Howdeshell et al. (2008) reported a NOEL of 1.5 µg EE2/kg BW/day, equal to

![Figure 2. MOEs (equal to the predicted estrogen intake from milk divided by the predicted estrogen intake from drinking water) for a young child. For E1, E2, and E3, MOEs are shown for exposure to prescribed estrogens predicted to be in drinking water and for naturally occurring estrogens predicted to be in drinking water. A single MOE is shown for EE2 because the only source of EE2 in drinking water is assumed to be therapeutic use (i.e., prescribed). MOEs for E1, E2, and E3 are based on the mass-based concentration of each estrogen in drinking water and milk. The EE2 MOE is based on the E2-eq concentration of EE2 in drinking water and of E1, E2, and E3 combined in milk.](image)

Table 2. MOEs for a child and an adult.

<table>
<thead>
<tr>
<th></th>
<th>Child</th>
<th>Adult</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Estrogen</strong></td>
<td><strong>Drinking water intake</strong></td>
<td><strong>MOE</strong></td>
</tr>
<tr>
<td></td>
<td>(µg/person-day)</td>
<td></td>
</tr>
<tr>
<td>Prescribed endogenous</td>
<td>8.2 × 10^-7</td>
<td>3,200</td>
</tr>
<tr>
<td>Naturally occurring endogenous</td>
<td>6.6 × 10^-7</td>
<td>400</td>
</tr>
<tr>
<td>Prescribed synthetic</td>
<td>1.0 × 10^-7</td>
<td>280</td>
</tr>
<tr>
<td>Total prescribed</td>
<td>1.1 × 10^-7</td>
<td>240</td>
</tr>
<tr>
<td>Total estrogens from all sources</td>
<td>1.7 × 10^-7</td>
<td>150</td>
</tr>
</tbody>
</table>

*Expressed as E2-eq.* *Compared with a child’s milk intake of 2.6 × 10^-4 µg/person-day (expressed as E2-eq).* *Compared with an adult dietary intake of 2.3 × 10^-7 µg/person-day (expressed as E2-eq).*

Exposure to estrogens in drinking water
90 µg EE2/person/day, and Tyl et al. (2008) reported a NOEL of 1 µg E2/kg BW/day, equal to 60 µg E2/person/day, assuming a body weight of 60 kg. Latendresse et al. (2009) summarized results from National Toxicology Program (NTP) studies of chronic and multigenerational reproductive effects of EE2. These authors observed an overall no observed adverse effect level for reproductive effects of 0.7 µg/kg/day in the five-generation reproduction study. Both the chronic and reproductive studies summarized by Latendresse et al. (2009) reported the same estrogen-specific ADIs because the estrogen-specific ADIs embody differences in activity). Compared to OELs and Australian guidelines, these ADIs are adjusted because the estrogen-specific ADIs embody differences in activity.

Figure 3. MOSSs for adult exposure to estrogens via drinking water for the WHO ADI, the TTC, four ADIs derived from OELs, and the four ADIs used to derive the Australian guidelines (EPHC et al. 2008). For the WHO ADI and the TTC, five MOSSs are presented corresponding to five categorizations of estrogens predicted to be in drinking water. MOSSs for the WHO ADI and TTC are based on estrogen intakes expressed as E2-eq (i.e., are activity adjusted), as are the MOSSs for total prescribed and total all sources comparisons to OELs and Australian guidelines, whereas MOSSs for the individual estrogens for OEL and Australian guideline comparisons are based on estrogen intakes expressed on a mass basis (i.e., are not activity adjusted because the estrogen-specific ADIs embody differences in activity).

Documenting that the potential exposure to total estrogens (prescribed and naturally occurring) in drinking water is at least 82 times lower than our natural background dietary exposure suggests that exposures to estrogens (prescribed or naturally occurring) in drinking water are inconsequential and should have no effect. Indeed, it is likely that naturally occurring day-to-day variation in dietary intake (e.g., having a glass of milk or some cheese one day and not the next) will lead to much larger variations in estrogen exposure than is associated with drinking water intake of prescribed estrogens or naturally occurring estrogens.

Similarly, the large MOSSs that result when predicted drinking water exposures are compared with several toxicity-based benchmarks are also an important finding and indicate that predicted drinking water exposures to estrogens are not expected to be associated with adverse effects in the U.S. population. Although there is no consistent approach for applying safety factors for infants and children or other sensitive subgroups, the combined safety factor applied to the toxicity-based benchmarks for interspecies variability and protection of sensitive subgroups adequately addresses issues associated with potential exposure of the developing fetus, infants, and children. Application of an additional safety factor of 10, as done in this analysis, is a conservative approach, as borne out by Latendresse et al. (2009) in the summary of the NTP studies. It is also consistent with the Food Quality Protection Act of 1996, which applies a default safety factor of 10 for sensitive subpopulations when dealing with pesticides in food products. Given that some of the ADIs we used in the present analysis are derived to be protective of sensitive subpopulations, such as prepubescent boys, the conclusion of an absence of an effect should extend to all segments of the U.S. population.

Even though the MOSSs resulting from all of the comparisons of estrogens in drinking water are consistently large, the question remains whether the uncertainty associated with the present analysis could lead to a substantially lower MOE or MOS. An underestimate of the MOE and MOS associated with prescribed and naturally occurring endogenous estrogens seems unlikely for several reasons. Perhaps most important is that we did not account for potential removal of estrogens by drinking water treatment plants. Few studies have examined such removal. However, in a recent study Benotti et al. (2009) reported drinking water treatment system removal rates of at least 80% and as much as 99% for E1, E2, and EE2 (E3 removal was not measured). In the present study, incorporating these removal data would have reduced predicted drinking water exposures (and, therefore, increased MOE and MOSs) by at least 5-fold and likely more.
MOEs derived from the dietary comparisons would also have been larger if concentrations of naturally occurring endogenous estrogens were available for all the foods in our diet and if the contribution of phytoestrogens to the estrogen content in food had been included.

MOEs and MOSs may be overestimated for some drinking water systems. The arithmetic mean is equal to approximately the 80th percentile of the distribution of drinking water PECs (Figure 1, Table 1). That means approximately 20% of drinking water systems included in PhA TE have critical low-flow PECs greater than those used in this analysis. The maximum critical low-flow drinking water PEC is about 10 times greater than the arithmetic mean PEC (Figure 1). The MOEs and MOSs corresponding to the maximum PEC would be about 10 times smaller than those associated with the mean PEC used in this analysis. However, even if we use the maximum PEC generated by PhA TE, prescribed estrogens predicted to be in drinking water still represent a fraction of background dietary estrogen exposure and remain below ADIs; thus, adverse effects are not expected.

MOEs and MOSs for estrogen exposure from all sources in drinking water may also be reduced in situations where drinking water system source water contains endogenous hormones from upstream animal husbandry operations or from pharmaceutical manufacturing. Such operations are not included in PhA TE because information on the location and magnitude of such sources is not readily available for most watersheds. The effect on most drinking water PECs of not including animal husbandry sources is expected to be small, given that most drinking water systems are likely to be near populated areas of watersheds and such areas are less likely to have animal husbandry operations than more remote areas of the watersheds. The effect on most drinking water PECs of not including pharmaceutical manufacturing sources is also expected to be small given that most pharmaceutical manufacturing sources provide onsite wastewater treatment and/or discharge to POTWs.

MOEs and MOSs for total estrogenic activity in drinking water may also be lower than shown by this analysis because compounds other than the endogenous and prescribed hormones that are the focus of this study may be present in POTW effluents. A comprehensive assessment of the potential effect of these other compounds is beyond the scope of this analysis. However, several studies investigating the overall estrogenic activity of POTW effluents report that most estrogenic activity is attributable to E1, E2, EE2, and EE2 (Aerni et al. 2004; Desbrow et al. 1998; Houtman et al. 2007; Salste et al. 2007). These results suggest that even if the estrogenic potential of the other compounds present in POTW effluents were included, the MOEs and MOSs estimated by this analysis would decrease by no more than about 2-fold.

One potential group of people who may have lower surface-water–related MOEs and MOSs than estimated here are anglers who consume self-caught freshwater fish. A bioconcentration factor of 635 has been reported for EE2 (Länge et al. 2001). Assuming that rate of bioaccumulation is representative of EE2 bioaccumulation in natural waters, an angler consuming 6.5 g of fish from the same surface water that serves as the source of his or her drinking water supply is exposed to as much EE2 through the consumption of fish as is present in 4.1 L of water, or about 2.25 times more daily exposure than he or she gets from drinking water. Although this does not alter the conclusions of this analysis because the MOEs and MOSs remain very large, it does indicate that, for some people, potential exposure to estrogens via consumption of fish may result in greater exposure than from consumption of drinking water.

Conclusion

The large MOEs and MOSs determined in the present study appear robust and are more likely to understate than overstate actual MOEs and MOSs. The dietary comparison indicates that potential exposures to trace levels of total estrogens (whether from a prescribed or naturally occurring source) predicted to be in drinking water in the United States are at least 82 times lower than exposures from background concentrations of naturally occurring estrogens in the diet. Drinking water exposures are also at least 28 times less than ADIs developed to be protective of sensitive populations. Taken together, the finding of consistently large MOEs and MOSs across all lines of evidence strongly suggests that concentrations of estrogens (including prescribed estrogens) predicted by PhA TE to potentially be in drinking water are not causing adverse effects in U.S. residents, including sensitive subpopulations.

Correction

In the original manuscript published online, some exposure estimates and margins of safety were incorrect. They have been corrected here.

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