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The importance of upland flow paths in determining urban effects on stream ecosystems

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Abstract. Mitigation of urban effects on streams requires an understanding of the paths by which urban effects are transmitted from catchments to streams and how those effects are attenuated with distance. We assessed whether modeling attenuation from impervious surfaces and septic tanks along drainage lines improved prediction of 3 instream ecological indicators. Eleven regression models were calculated for each indicator (*Escherichia coli*, NO_3/NO_2, and Stream Invertebrate Grade Number Average Level [SIGNAL; a macroinvertebrate assemblage composition index]). Predictor variables included imperviousness or septic-tank density with no attenuation (i.e., total imperviousness and tank density), with overland attenuation (exponential decay with distance along topographic flow paths to stormwater drain or to stream), and with overland and instream attenuation (exponential decay with distance travelled within the stream). *Escherichia coli* was best predicted by the weighted density of septic tanks, with their influence attenuated to near 0 within tens of meters of the stream and within thousands of meters along streams. These results suggest strong overland attenuation of bacterial contamination and indicate that stormwater drains are not important pathways for septic leakage in this area. NO_3/NO_2, which is mobile through soils, was best predicted by nonattenuated septic-tank density within catchments. SIGNAL was best predicted by either impervious area or septic tanks within tens of meters of stormwater drains, a result suggesting that SIGNAL is most strongly affected by stormwater runoff routed through stormwater pipes to streams. Management of dry-weather fecal and N contamination in the study streams should focus on septic tank management (stricter maintenance regimes or replacement with other sewage management systems). Near-stream septic tanks are probably most important for fecal contamination, whereas catchment-wide management probably will be required for reduction of dry-weather N concentrations. Management of broader instream ecological condition, as indicated by macroinvertebrate assemblage composition, probably will require catchment-wide retention of stormwater runoff (using tanks or infiltration systems) to mimic natural flow-paths between catchments and streams.

Key words: urbanization, stormwater, septic tank, stream ecology, flowpath, watershed, regression modelling, distance weighting.

Mitigation and, potentially, prevention of negative effects of urban land use on stream ecosystems requires an understanding of the nature of instream stressors driving urban stream degradation, the catchment sources of those stressors and the pathways by which the stressors are transmitted between catchments and streams (Wenger et al. 2009). Much of the urban fabric that might contribute to stream degradation, such as the impermeability of its roofs and roads, is unlikely to be significantly altered under any feasible management transformation, but hydrologic pathways in cities have greater potential for redesign to reduce urban effects on receiving aquatic environments. Therefore, study of upland flow paths is critical to the discipline of urban stream ecology and to urban stream management.

The effects of land use on stream ecological condition might be influenced by the spatial configuration of each landuse type and its proximity to the

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stream channel (King et al. 2005, Van Sickle and Johnson 2008). This influence probably is mediated through transmission and attenuation of effects along hydrologic pathways. Vegetated riparian zones can be of particular importance in mitigating catchment landuse effects because they lie at the interface between terrestrial and stream environments (Naiman and Décamps 1997). Riparian vegetation and soils attenuate landuse effects by intercepting and retaining potential pollutants and preventing their transport to streams (Lowrance et al. 1984). Similar attenuation processes are possible along upland drainage lines and potentially could be mimicked by engineered retention structures, such as wetlands or biofiltration/infiltration systems (Walsh et al. 2005a).

In modelling studies, prediction of instream ecological response was improved if disturbed land use was weighted by distance along topographic flow paths (King et al. 2005, Van Sickle and Johnson 2008). Such models suggest that landuse effects on stream ecosystems can be mitigated if flow from potentially damaging land uses can be intercepted along topographic drainage lines by undeveloped, nonagricultural land, such as forests. However, urbanization is characterized by construction of stormwater drainage networks that bypass natural topographic flow paths and are designed for hydraulic efficiency. Thus, less attenuation of hydrologic and water-quality effects is more likely along stormwater drains than along natural topographic flow paths. Soils and vegetation of topographic drainage lines provide opportunities for retention or loss of water through infiltration and evapotranspiration, processes that serve to diminish hydraulic or water-quality effects from urban stressors, such as stormwater runoff from impervious surfaces or seepage from septic tanks. The capacity for attenuation of effect is likely to increase with length traveled along a topographic drainage line. In contrast, the hydraulic efficiency of stormwater drainage via constructed pipes and channels probably will convey such effects efficiently with little or no attenuation to a stream. This contrast in hydraulic efficiency between overland flow paths and stormwater drains underlies the concept of effective imperviousness (EI; Shuster et al. 2005), which predicts much stronger effects from impervious surfaces that drain to stormwater drains than from those that drain to pervious drainage lines.

Our goal was to assess the importance of different flow paths on attenuation of the effect of different urban stressors on stream ecosystems across a range of urban, exurban, and rural streams on the eastern fringe of Melbourne, Australia. We quantified the catchment density of 2 landscape features that have been implicated as contributors to urban effects on streams: 1) septic tanks, which are potential sources of fecal and nutrient contamination (e.g., Victorian Auditor-General's Office 2006), and 2) impervious surfaces, which produce stormwater runoff that is widely implicated as a major cause of many ecological changes observed in urban streams (Walsh et al. 2005b). For each septic tank and impervious surface, we quantified flow distances to instream sampling sites along 2 overland flowpaths: 1) along topographic drainage lines of the natural catchment and 2) along topographic drainage lines minus the lengths intercepted by the constructed stormwater network. We assessed the responses of 3 instream variables, *Escherichia coli* and NO_3/NO_2 concentrations, and macroinvertebrate assemblage composition (variables often targeted by urban stream managers) to these 2 potential sources of stress along the 2 flowpaths. We demonstrated that each of these variables probably is influenced primarily by different catchment-scale causal agents with different degrees of attenuation along different hydrologic pathways. These differences underscore the importance of linkages between urban catchment hydrology and stream ecology and have important implications for prioritization of management actions for conservation or restoration of urban streams.

Methods

Study sites and sampling

We used stream-monitoring and landuse data from several studies from the eastern fringe of the Melbourne metropolitan area, where land use ranges from medium-density urban to mixed agriculture and forested reserve. The spatial extent of the study area was limited by available data for locations of septic tanks. Thirty-three sites (of which 17 were independent, in that they had no sites upstream) were sampled for *E. coli* and NO_3/NO_2 in streams with catchments ranging from 1 to 89 km^2 (Fig. 1).

Escherichia coli and NO_3/NO_2 data were from 3 catchment-specific studies (McGuckin 1998, Pettigrove and Coleman 1999, Frame et al. 2005), each conducted over 1 y during the period from 1997 to 2003. We also collected samples during an additional sampling campaign in 2007. Median concentrations of *E. coli* and NO_3/NO_2 were determined from 7 grab samples taken every 2 to 4 wk over 2 to 5 mo, and log(x)-transformed median values were used as response variables. All analyses were done with standard methods in accredited laboratories (National Association of Testing Authorities; http://www.nata.asn.au). NO_3/NO_2 samples were filtered (0.2 μm) into clean polyethylene bottles in the field and frozen until

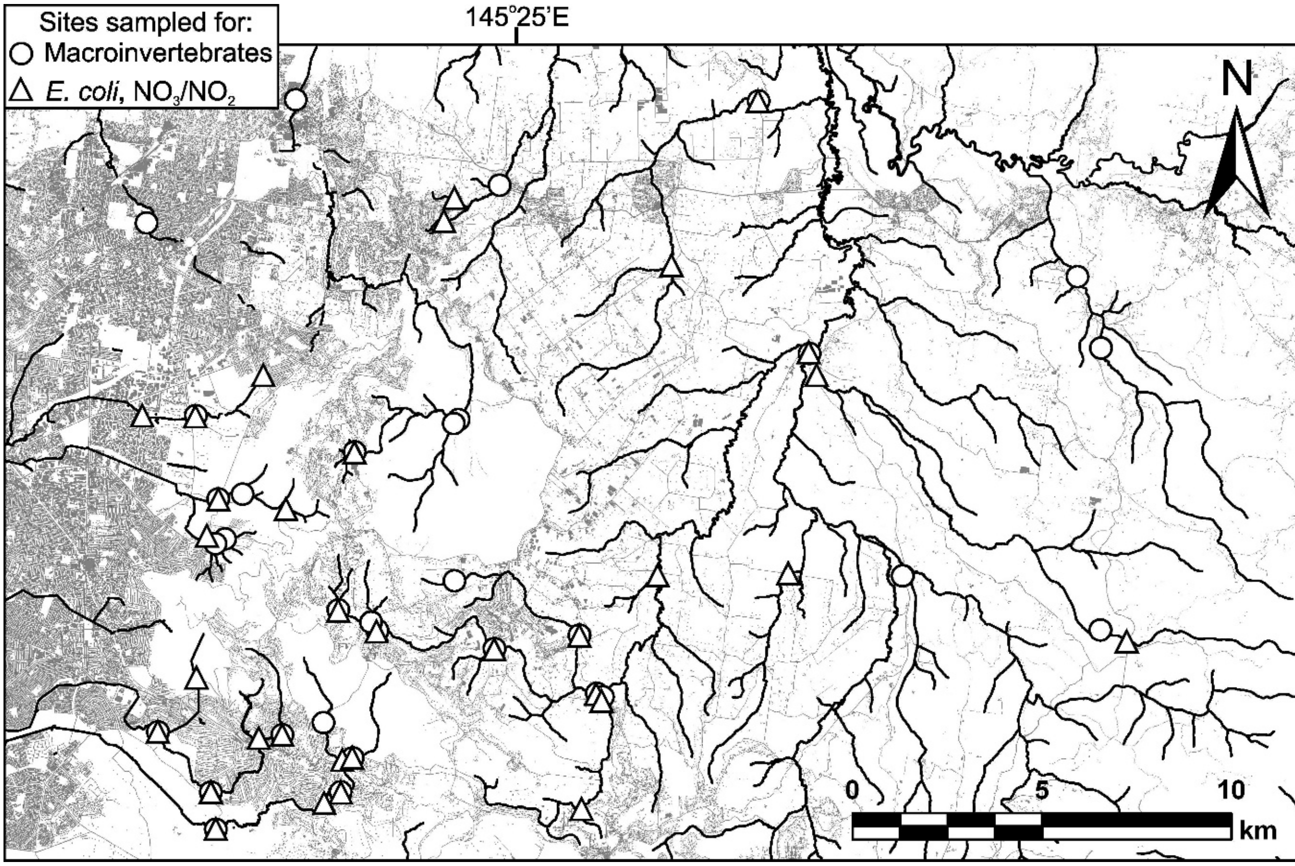

FIG. 1. Sampling sites for *Escherichia coli*, NO_3/NO_2, and macroinvertebrates on the eastern fringe of the Melbourne metropolitan area. Light gray polygons indicate impervious areas.

laboratory analysis (APHA 1998). *Escherichia coli* concentrations were determined by membrane filtration (McGuckin 1998, Pettigrove and Coleman 1999, and the 2007 campaign) or the Colilert® (Idexx Laboratories, Westbrook, Maine) technique (Frame et al. 2005).

To assess the extent to which variation among sites sampled in different years might be explained by interannual variation, we resampled 3 sites with low, moderate, and high median *E. coli* concentrations (261, 972, and 1916 *E. coli*/100 mL, respectively) in the earlier studies during the 2007 sampling campaign. Variance of median log(x)-transformed *E. coli* concentrations among years was 6 to 14% of variance among the 3 sites, a result suggesting that median *E. coli* values were relatively stable, and thus, were robust measures of baseflow concentrations over the study period. Interannual variance in median log(x)-transformed NO_3/NO_2 values was more variable (0–62% of intersite variance), a result suggesting that temporal variation could be a large source of error in models for this variable.

We extracted macroinvertebrate data from the Melbourne Water (MW; http://www.melbournewater.com.au) macroinvertebrate database (MW, unpublished data, but data from 12 sites were used by Walsh et al. 2005a). Macroinvertebrates were sampled from 80- to 100-m reaches at 32 sites (18 also were sampled for *E. coli* and NO_3/NO_2). Nineteen of the 32 sites were independent (Fig. 1). All samples were collected between 2000 and 2005 with rapid bioassessment methods: qualitative kick-sampling of riffles and sweep sampling of edge habitats with a standard 250-μm-mesh dip net, followed by standardized sorting of live animals in the field (Tiller and Metzeling 1998), and identified to family. Presence/absence data from 4 samples (riffle and edge, spring and autumn, collected over 1 y) were combined for each site. Stream Invertebrate Grade Number Average Level (SIGNAL) score, an index based on tolerance for organic pollution of macroinvertebrate families (Chessman 2003), was used as the response variable. SIGNAL is a strong correlate of assemblage compositional similarity and of degree of urban effect among sites in the region of our study (Walsh 2006). In this region, a SIGNAL score >6 indicates good ecological condition similar to streams in undeveloped forested catchments, whereas a score <5 indicates a degraded condition similar to metropolitan streams of Melbourne (Walsh 2006).

Spatial data

We used an inventory of septic tanks for the study area assembled by the Shire of Yarra Ranges in 2002 (http://www.yarraranges.vic.gov.au/). Each of the ~20,000 tanks was associated with a property and mapped at the centroid of a property parcel of the Victorian cadastre (http://www.land.vic.gov.au/). A small portion of the southern study area fell outside the septic tank inventory area. For this rural, unsewered area, we assumed that each house had a septic tank, and we manually digitized ~900 house locations from aerial photographs (MW; 2001) for use as inferred locations of septic tanks. Septic tanks usually are close to houses, so the location of a house is likely to be a more accurate estimate of septic tank position than the property centroid used in the Shire of Yarra Ranges inventory.

We mapped impervious areas with spatial integration of digital road network and local government building area data, aerial orthophotographs, and ground-truthing across the study area (see Taylor et al. 2004).

We developed 2 digital elevation models (DEMs) with 10-m contour data from the Victorian 1:25,000 digital topographic map series (http://www.land.vic.gov.au/) and a geographic information system (GIS) layer (natural waterway centerline) that details the streams of the region managed by MW. We ground-truthed the extent of perennial streams in the region through many observations throughout the study period and made minor adjustments to the MW stream layer to produce a GIS layer of perennial streams. We conditioned a ground DEM derived from the contour data alone in 2 ways to develop 2 hydrologically compatible DEMs. We conditioned the 1st DEM to the perennial stream network and used it to portray natural topographic flow paths. We conditioned the 2nd DEM to the perennial stream network and stormwater drainage pipes/lined drains (local government GIS data of stormwater drainage pipe lines, accessed through MW) and used it to portray flow paths to and through the stormwater drainage network (for details of DEM methods, see Kunapo et al. 2009).

We assigned each septic tank or impervious surface 3 distance attributes calculated from the DEMs with distributed hydrologic modeling tools (ESRI 2004): 1) overland flow distance to stream (d_L), 2) overland flow distance to drain (d_D), and 3) instream flow distance to the sampling site (d_W; Fig. 2). d_L was the overland distance along the natural topographic flow path to the nearest stream from each septic tank or from the most downstream point of each impervious surface. If the flow path did not cross a stormwater drain before reaching the stream, then $d_D = d_L$. However, if the flow path did cross a stormwater drain, then d_D was the distance from the tank or the impervious surface to the drain (models that use d_D assume no attenuation through stormwater drains). We attributed every tank or impervious surface to a subcatchment (Fig. 2), and d_W was the stream distance to the sampling site from the most downstream point of that subcatchment. We used the stormwater DEM to derive subcatchment boundaries on each stream for confluence points where the stream met tributaries of any size or major stormwater pipes and for sampling sites (Fig. 2). Where the stream distance between subcatchments was >250 m (in a few rural reaches), we created additional subcatchments at a maximum of 250 m. Thus, the maximum possible error for d_W was 250 m.

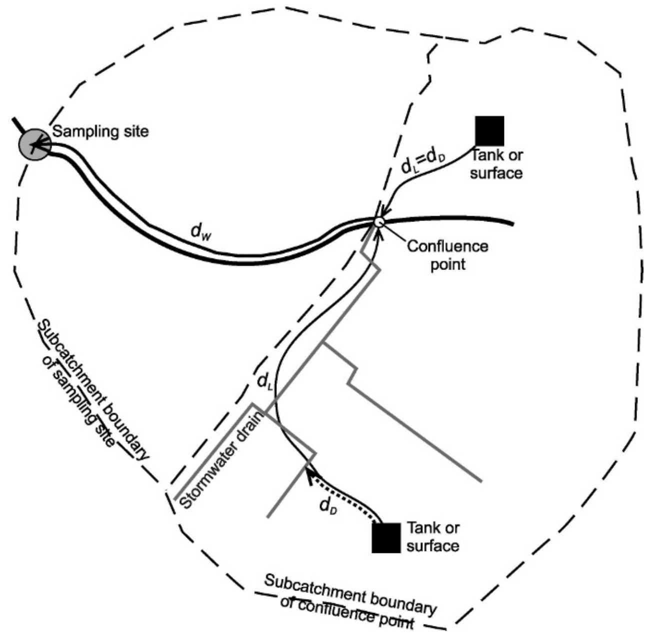

FIG. 2. Flowpath distances from a septic tank or the most downstream point of an impervious surface to a stream sampling site. d_L is the natural topographic flow path to the stream, whereas d_D is that portion of d_L before the flowpath intersects a stormwater drain or pipe. $d_L = d_D$ where no stormwater pipes exist. The instream flow distance (d_W) is measured from the confluence point to the stream sampling site.

Statistical models

Our modeling approach followed that of Van Sickle and Johnson (2008) and Johnson et al. (2007). This approach is based on the assumption that the influence of a land use at a point in the catchment is a nonincreasing function of the flow path between that point and the point at which the stream response variable is sampled. Van Sickle and Johnson (2008) compared 3 attenuation functions to describe the decline of influence along flow paths. They found little difference in goodness-of-fit between exponential decay, inverse distance, and threshold functions (the first 2 predicted a continuous decline in influence with distance, the last predicted no influence beyond a threshold distance). Van Sickle and Johnson (2008) recommended using ecological and physical criteria rather than statistical criteria when choosing attenuation functions. We chose the exponential decay function to model attenuation of effect from septic tanks and impervious surfaces because it is appropriate for weighting landuse effects on contaminants that are progressively depleted along flow paths (Johnson et al. 2007, Van Sickle and Johnson 2008).

We modeled 3 combinations of factors that reflect alternative assumptions about stressor sources and pathways: 1) impervious surfaces or septic tanks within catchments as stressor sources, 2) overland topographic drainage lines or perennial streams as attenuation pathways, and 3) overland flow path to the stream either entirely along a topographic drainage line (d_L) or intersected by a stormwater drain, through which flow to the stream is hydraulically efficient, with no attenuation (d_D). We tested models that included a subset of combinations of these factors (Table 1) and determined the goodness-of-fit for a range of parameter values for each model. The most complex models included both instream and overland attenuation. As the length of attenuation approaches infinity, attenuation models approach equivalence to simpler, unattenuated models, which are parsimoniously preferable. Therefore, in addition to models with both instream and overland attenuation, we separately assessed models with overland, but not instream, attenuation and models with no attenuation along either flowpath. Overland attenuation is likely to be stronger than instream attenuation (Van Sickle and Johnson 2008), so we did not consider models with instream attenuation and no overland attenuation. We did not assess more complex combinations of the 3 factors because of the limitations of our data (see below).

For median *E. coli* concentration, median NO_3/NO_2 concentration, and macroinvertebrate SIGNAL score, we tested 5 flowpath attenuation functions for each of the 2 stressor sources (septic tanks and impervious surfaces). For each of the septic tank models, the independent variable derived for each sampling site was attenuated septic-tank density (S; weighted no.

TABLE 1. Model quality of the best-fitting regressions of each model type for median *Escherichia coli* and NO_3/NO_2 concentrations and for the macroinvertebrate Stream Invertebrate Grade Number Average Level (SIGNAL) index in streams of eastern Melbourne. For each model type, the 3 factors being assessed (stressor source [S = septic density, I = imperviousness], overland flow attenuation type, and instream flow attenuation) are described and the number of parameters used (K) to calculate the Akaike Information Criterion adjusted for small sample size (AIC_c) is listed. K = 3 for a simple regression with 1 predictor variable because it includes the residual squared error. The difference between AIC_c for each model and that of the best model (ΔAIC_c) is indicated for each variable. For each response variable, * indicates R^2 for the best (ΔAIC_c = 0) or equally plausible (ΔAIC_c < 2) models.

Model code	Stressor and attenuation weighting	Attenuation models			*E. coli*		NO_3/NO_2		SIGNAL	
		Overland	Instream	K	ΔAIC_c	R^2	ΔAIC_c	R^2	ΔAIC_c	R^2
a	S unweighted	None	No	3	26.0	0.00	**0.0**	**0.34***	30.1	0.00
b	I unweighted	None	No	3	24.9	0.03	11.6	0.07	26.3	0.12
c	$S + I$ unweighted	None	No	4	23.5	0.14	2.2	0.35	21.8	0.29
d	S overland to stream	To stream	No	4	3.3	0.54	2.9	0.34	11.2	0.49
e	I overland to stream	To stream	No	4	18.1	0.27	14.8	0.05	19.1	0.35
f	S overland to drain	To drain	No	4	12.3	0.39	3.0	0.34	**0.0**	**0.64***
g	I overland to drain	To drain	No	4	22.1	0.18	14.9	0.05	**1.4**	**0.63***
h	S overland to stream, instream to site	To stream	Yes	5	**0.0**	**0.61***	5.6	0.34	14.1	0.49
i	I overland to stream, instream to site	To stream	Yes	5	19.5	0.30	17.7	0.05	21.9	0.35
j	S overland to drain, instream to site	To drain	Yes	5	11.1	0.46	5.7	0.34	2.9	0.64
k	I overland to drain, instream to site	To drain	Yes	5	23.5	0.21	17.7	0.05	4.3	0.63

tanks/area of catchment).

$$S = \frac{\sum_i W_i}{A_C} \quad [1]$$

where W_i = weighting applied to the i^{th} tank, and A_C = catchment area. For each of the impervious surface models, the independent variable was attenuated imperviousness (I; weighted impervious area as a percentage of catchment area).

$$I = 100\left(\frac{\sum_j (A_j W_j)}{A_C}\right) \quad [2]$$

where A_j = the area of the j^{th} impervious surface, W_j = the weighting applied to A_j. The 5 weighting functions were:

i) Unweighted S or I ($W = 1$), with neither overland nor instream attenuation (models a, b, and c; Table 1).

ii) S or I attenuated overland to stream (models d and e; Table 1).

$$W = e^{\left(\frac{-d_L}{\alpha}\right)} \quad [3]$$

where α = overland attenuation parameter. (Note that exponential decay parameters reported here are reciprocals of those reported by Van Sickle and Johnson 2008.) This function assumes attenuation along topographic drainage lines over land to the stream, but no instream attenuation.

iii) S or I attenuated overland to drain (models f and g; Table 1). This function has the same form as [3], but uses d_D instead of d_L and assumes attenuation along topographic drainage lines over land until it intersects a stormwater drain or a stream, but assumes no attenuation along the drain or the stream. In the few cases where the impervious surfaces of an upland urban area drained to stormwater pipes, but these pipes, in turn, drained to an upland part of the catchment, we summed the attenuated effects of tanks to the stormwater drains and then calculated attenuation of the summed effect for the overland flow path between the outlet pipe and the nearest stream.

iv) S or I attenuated overland to stream and instream to site (models h and i; Table 1).

$$W = e^{\left(\frac{-d_L}{\alpha}\right)} e^{\left(\frac{-d_W}{\beta}\right)} \quad [4]$$

where β = instream attenuation parameter. This function assumes overland flow attenuation as in function ii, followed by attenuation at a potentially different rate along the stream to the sampling site.

v) S or I attenuated overland to drain and instream to site (models j and k; Table 1). This function has the same form as [4], but uses d_D instead of d_L and assumes overland flow attenuation to a stormwater drain as in function iii, followed by attenuation at a potentially different rate along the stream to the sampling site.

For each of the 3 stream response variables, we assessed 5 regression models, one for each weighting function, with I as a predictor and with S as a predictor. We also assessed an 11^{th} regression model with 2 predictor variables, unweighted S and unweighted I (model c; Table 1).

Weighting S or I had the potential to alter their statistical distributions with the consequence that some sites could become outliers that might apply excessive leverage to the regression and produce spuriously good fits (Quinn and Keough 2002). We guarded against this possibility by assessing the effect of different weighting schemes and parameter values on the distributions of S and I. For I, $\log(x + 0.1)$-transformation prevented outliers (assessed with boxplots) and was applied for all weighting functions. For S, $\log(x + 0.1)$-transformation produced outliers under schemes with little or no weighting, whereas untransformed S produced outliers under heavily weighted schemes. Therefore, we confirmed that this behavior of the data did not affect our inferences by conducting all regressions on S twice, once with $\log(x + 0.1)$-transformed data and once with untransformed data.

For models that included the exponential decay parameters α or β, we searched for the best-fitting model by specifying a vector of values, spaced exponentially (1–10^4 m for α, 1–10^6 m for β; values exceeded the full range of d_L and d_W values, respectively) or by a rectangular array of 2 parameters for weighting functions iv and v. We then found the best-fitting parameters for each weighting model by searching within a smaller range of more finely spaced parameter values in the region of best fit. Because exponential decay parameters (α and β) are difficult to interpret, we report half-decay distances (HDD = parameter multiplied by ln2). W is reduced by ½ over HDD and is reduced by ~97% over 5 × HDD.

We used the Akaike Information Criterion adjusted for small sample size (AIC_c) to assess the relative quality of alternative models (Burnham and Anderson 2002). The absolute value of the AIC_c is difficult to interpret. Therefore, we report ΔAIC_c, the difference between a model's AIC_c and that of the overall best-fitting model, so that the ΔAIC_c of the best-fitting model = 0. Models with lower AIC_c are better fits, and models with a difference in $AIC_c \leq 2$ are considered equally plausible (Burnham and Anderson 2002), but in such cases the simplest model (with fewest parameters) is preferable (Quinn and Keough 2002). A difference in AIC_c of 4–7 indicates that the model with the lower AIC_c is superior, whereas a difference in $AIC_c > 10$ indicates that the model with the lower AIC_c is strongly preferred (Burnham and Anderson 2002). We report R^2 values of regressions as an indication of overall model quality.

We restricted our analyses to linear models of response variables. Walsh et al. (2005a) modeled nonlinear responses of instream ecological variables and found a linear decline in ecological condition with increasing EI to a threshold beyond which no further degradation occurred. We sampled only sites with low levels of catchment urbanization (EI $\leq$ 16%) within the range of EI for which linear models are likely to be appropriate (Walsh et al. 2005a).

We did not model directly within-network dependencies for our stream response variables, but we were careful to avoid potentially over-parameterized models in the possible presence of spatial autocorrelation. Some of the sites in our study fall along the same drainage line, but they all are separated by tributaries, and stream distances between them range from 2 to 30 km. Thus, distance and tributary effects probably reduced spatial autocorrelation in all 3 response variables among the sites. The effective degrees of freedom for our analyses are probably 20 to 30 (32–33 observations, but only 17–19 independent subcatchments). Tabachnick and Fidell (1996) recommended $\geq$10 observations for each predictor variable in regression models. Our most complex models use 3 predictor variables. Thus, these models probably are near the limit of parameterization for our data. However, the models include instream attenuation (β), which would have the effect of further reducing downstream dependencies among observations and increasing effective degrees of freedom. Thus, we interpret the quality of fit of these most complex models (h–k; Table 1) with caution.

Results

Unweighted *S* was approximately normally distributed among the sampling sites, and ranged from 0 to 135 tanks/km^2. The relative plausibility of models for each response variable was unchanged if *S* was untransformed or log(x + 0.1)-transformed, but to minimize leverage of outliers in the final models, log(x + 1)-transformed data are presented for *E. coli*, and untransformed data are presented for NO_3/NO_2. Unweighted *I* ranged from 0.2 to 18%. The distribution and range of each variable was similar for the 2 sets of sampling sites used in analyses (Table 2), a result suggesting that differences in catchment characteristics between the 2 data sets are unlikely to explain differences in modelling results among the 3 response variables.

TABLE 2. Mean and range unweighted septic tank density (*S*) and total imperviousness (*I*) for the 2 sets of data used in this study. The distributions of the 2 stressors were similar for sites sampled for *Escherichia coli* and NO_3/NO_2, and those sampled for macroinvertebrates.

Data set	*S*	*I*
E. coli and NO_3/NO_2	62 (1.5–135)	6.2 (0.2–18)
Macroinvertebrates	61 (0–135)	5.7 (0.2–18)

Escherichia coli concentration was poorly predicted by unweighted *S* and by unweighted *I* (models a and b; Table 1). It was best predicted by *S*, attenuated overland to a stream (*HDD* 4.3 m), and then attenuated instream (*HDD* 3.8 km; Table 1, Fig. 3A, B). The same model structure (model h; Table 1) was equally plausible ($\Delta AIC_c \leq 2$) for overland *HDD* of 0–7.3 m and for instream *HDD* of 2.0–10.3 km (Table 3, Fig. 3B). A less complex model, without instream distance weighting was marginally less plausible (model d; Table 1). Models with overland attenuation to stormwater drains were substantially less plausible (models f and j; Table 1). Thus, these analyses suggest that *E. coli* contamination from septic-tank leakage or overflow is attenuated within tens of meters along topographic flow paths in our study catchments (*HDD* of 7.3 m = a 97% reduction of effect at 47 m), and over thousands of meters of stream flow (97% reduction of effect at 10–52 km).

Median NO_3/NO_2 concentrations were best predicted by unweighted *S* (model a), but the relationship was not strong (R^2 = 0.34; Table 1, Fig. 4). The single high outlier in the regression might have been influenced by agricultural runoff because the sampling point was downstream of several farms. If this point were omitted, the relative plausibility of competing models would be unchanged and the fit of the unweighted septic-tank density model would be stronger (R^2 = 0.43). No attenuation model improved the goodness-of-fit enough to produce a more plausible regression model. The possibility of some attenuation of NO_3/NO_2 over long distances cannot be discounted, but attenuation over shorter distances (overland to stream *HDD* < 185 m, and

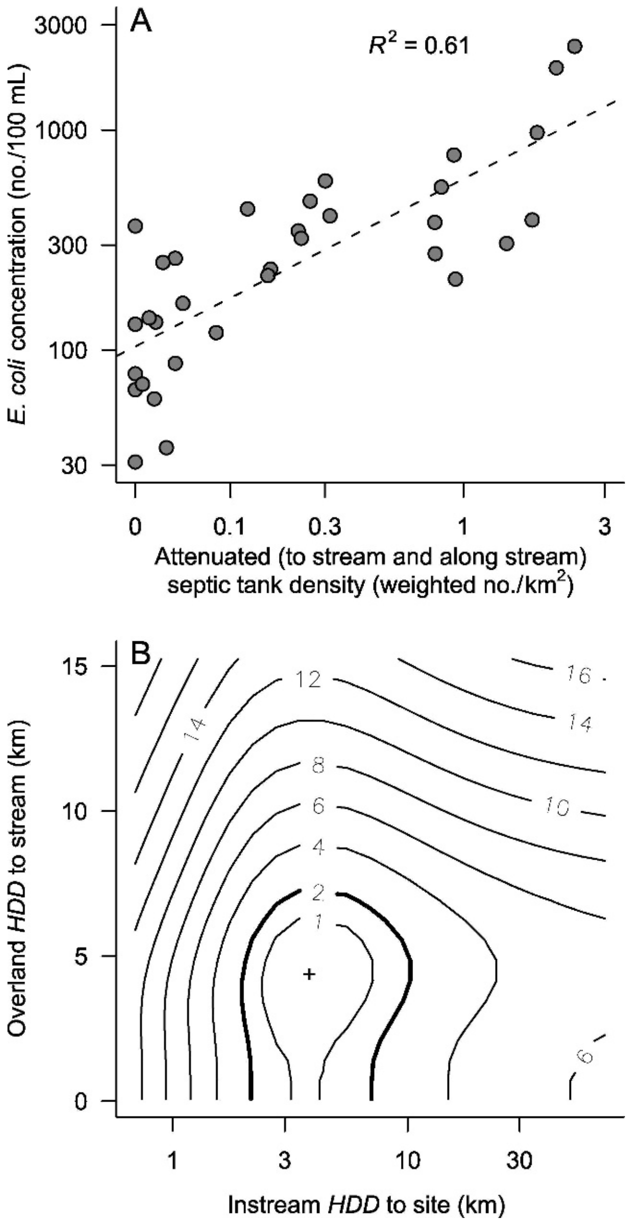

FIG. 3. A.—Best-fit regression model predicting *Escherichia coli* concentration in streams on the eastern fringe of the Melbourne metropolitan area. The best-fit regressor was the catchment density of septic tanks attenuated by an overland half-decay distance (*HDD*) to stream of 4.3 m and an instream *HDD* of 3.8 km. B.—Contour plot of difference in Akaike Information Criterion adjusted for small sample size from the best-fit model (ΔAIC_c) for the attenuated septic-tank density model over a range of overland (to the nearest stream) and instream *HDD*s. The best-fit *HDD*s ($\Delta AIC_c = 0$) are indicated as +, and *HDD*s with $\Delta AIC_c < 2$ (bold line) are equally plausible (i.e., overland *HDD* 0–7.3 m and instream *HDD* 2.0–10.3 km).

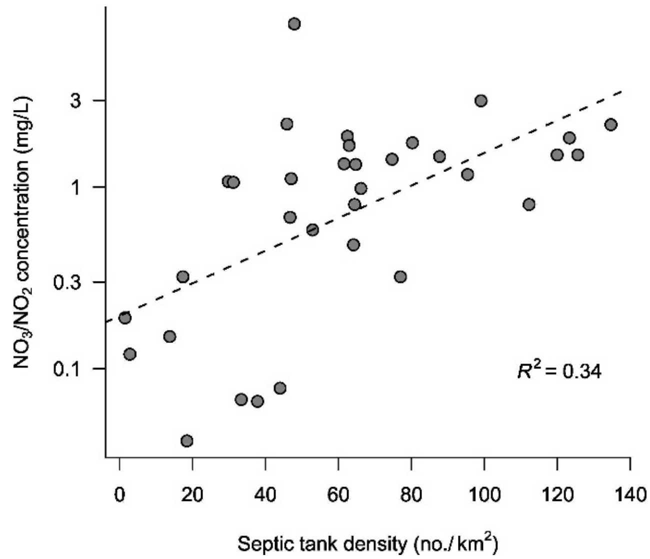

FIG. 4. Best-fit regression model predicting NO_3/NO_2 concentration in streams on the eastern fringe of the Melbourne metropolitan area (unattenuated septic-tank density as sole regressor).

instream *HDD* of < 8.5 km) is highly improbable (Table 3). Therefore, the position of septic tanks within our study catchments probably had little effect on median NO_3/NO_2 concentrations.

SIGNAL was much more strongly predicted by models with attenuation to stormwater drains than those with attenuation to streams or without attenuation (Table 1). SIGNAL was best predicted by 2 equally plausible models: *I* attenuated overland to stormwater drain (*HDD* 4.3 m, model g; Table 1, Fig. 5A), and *S* attenuated overland to stormwater drain (*HDD* 0.6 m, model f; Table 1, Fig. 5B). *HDD* values of 0–9.4 m were equally plausible for *I*, but the range of plausible *HDD* values for *S* was much shorter (0–2.2 m; Table 3, Fig. 5C). Models with instream attenuation (models j and k; Table 1) were less plausible, particularly for *HDD* < 3.4 km (Table 3). Models assuming overland flow to stream rather than to stormwater drain were substantially less plausible (models d and e; Table 1). These analyses suggest that macroinvertebrate assemblages of streams in our study are most strongly influenced by runoff from impervious surfaces or by septic-tank contamination transmitted through the stormwater drainage network rather than along topographic flow paths. The attenuation model suggests that the effect of either stressor is diminished to near 0 within tens of meters of overland flow along topographic flow paths (a 97% reduction of an impervious effect is predicted at 22 m for the most plausible *HDD* of 4.3 m).

TABLE 3. Half-decay distances (*HDD*) for median *Escherichia coli* and NO_3/NO_2 concentrations (stressor source: septic tanks) and for the macroinvertebrate Stream Invertebrate Grade Number Average Level (SIGNAL) index (stressor source: impervious surfaces or septic tanks) in streams of eastern Melbourne. The most plausible (best-fit) model for each variable had the lowest Akaike Information Criterion adjusted for small sample size (AIC_c). Ranges of *HDD*s that are equally plausible (models for which AIC_c exceeded AIC_c of the best-fit model by <2; i.e., $\Delta AIC_c < 2$) are shown in parentheses. The least plausible ranges of *HDD*s, beyond which models are clearly inferior to the best-fit model ($\Delta AIC_c > 10$), also are shown. Models with no attenuation (indicated by n/a) are computationally equivalent to attenuation models with infinite *HDD*, but because attenuation models are more complex than the no-attenuation model, all values of *HDD* result in $\Delta AIC_c > 2$ and are less plausible than simpler models. However, for such models, shorter *HDD* values are increasingly implausible.

Model	Overland *HDD* (m)		Instream *HDD* (km)	
	Most plausible	Least plausible	Most plausible	Least plausible
E. coli (Septic tanks as source, overland to stream)	4.3 (0–7.3)	>13.2	3.8 (2.0–10.3)	<0.8
NO_3/NO_2 (Septic tanks as source, overland to stream)	n/a	<185	n/a	<8.5
SIGNAL (Impervious surfaces as source, overland to drain)	4.3 (0–9.4)	>112	n/a	<4.9
SIGNAL (Septic tanks as source, overland to drain)	0.6 (0–2.2)	>6.2	n/a	<3.4

Discussion

The 3 stream response variables used in our study differed in the most likely catchment-scale causal agent of their variation, in the hydrologic pathway by which the effect was transmitted, and in the degree of attenuation along those flow paths. These differences demonstrate important linkages between urban catchment hydrology and stream ecology and have important implications for the prioritization of management actions for conservation or restoration of urban streams.

Our results suggest that natural upland flow paths can strongly attenuate stormwater and sewage effects in urban areas, except for contaminants, such as NO_3/NO_2, that are mobile through soils (Stumm and Morgan 1996). Septic-tank seepage is the most likely primary source of *E. coli* and NO_3/NO_2 in our study streams. These and other contaminants from septic-tank seepage are most likely transmitted to the stream along topographic flow paths (i.e., the most likely attenuation pathway for *E. coli* attenuation) rather than through stormwater drains. In contrast, macroinvertebrate assemblages (as indicated by SIGNAL score) were most probably degraded by catchment-derived stressors transmitted to the stream through stormwater drains, but these effects also were likely to be strongly attenuated along topographic flow paths before entering stormwater drains. The stormwater drainage system was the most likely pathway for transmission of the catchment stressors causing degradation of macroinvertebrate assemblages. Thus, stormwater runoff from impervious surfaces (perhaps compounded by septic-tank overflows through stormwater drains during floods) was the most likely primary source of stress, despite the fact that septic tanks and impervious surfaces were statistically equally plausible sources of stressors.

Effects transmitted through stormwater drains

The range of macroinvertebrate assemblage composition among the sites in our study was similar to that reported in an earlier study from the same region (Walsh 2004). Assemblages ranged from degraded with very few or no pollution-sensitive families (sites with attenuated imperviousness > 1%; Fig. 5A) to diverse with many sensitive families and similar to assemblages found in undisturbed catchments of the region (sites with very low attenuated imperviousness; Fig. 5A). Macroinvertebrate assemblage composition in streams of this region was strongly correlated with EI, as were other instream ecological indicators, such as diatom assemblage composition, algal biomass accrual, electrical conductivity, concentrations of filterable reactive P and dissolved organic C (Walsh et al. 2005a), and breakdown rate of exotic labile leaves (Imberger et al. 2008). Thus, the most plausible model predicting macroinvertebrate assemblage composition in these streams–imperviousness attenuated within tens of meters of stormwater drains–is also likely to be the most plausible predictor of broader shifts in instream ecological structure and function associated with urbanization in this region.

The distance-weighting model for imperviousness, objectively derived in our study as the best predictor of macroinvertebrate assemblage composition, is similar in form to the more subjectively derived method for determining EI used by Walsh et al. (2005a). In that study, only impervious surfaces that were within 20 to 40 m upslope of a stormwater drain were assumed to be connected to the drain, and therefore, counted as EI. Among our study sites, EI measured by the method of Walsh et al. (2005a) ranged from 0 to 65% of total imperviousness (i.e., 0–65% of impervious surfaces were considered connected to a drain), whereas attenuated imperviousness as calculated by the most plausible exponential decay model of our study ranged from 0 to 45% of total imperviousness. In analyses not presented here, we determined that the best-fit threshold distance upslope of stormwater drains for predicting SIGNAL score using a similar EI measurement was 11.5 m. However, this model was less plausible than the best-fit model using exponential decay weighting ($\Delta AIC_c = 6.1$), suggesting that attenuated imperviousness (calculated using exponential decay weighting) could be a better indicator than effective imperviousness, calculated using a threshold distance.

Exponential decay weighting of the overland parameter in the most plausible model probably portrays the decreasing likelihood that an impervious surface will be connected to a drain as distance from the drain increases. More important, the strength of the overland distance weighting model and the estimated magnitude of its weighting parameter suggest that impervious surfaces that are disconnected from the stormwater network and from streams by only tens of meters of pervious land have very little or no effect on stream ecological structure and function in our study area.

Thus, in those areas of our study catchments without stormwater drains, the effects of urban land use, such as stormwater runoff and septic-tank leakage, on macroinvertebrate assemblages (and probably other instream ecological indicators), are likely to be predominantly from sources very close to streams. In catchments with stormwater drainage networks, stormwater runoff from a much larger area, including upland areas, potentially contributes to

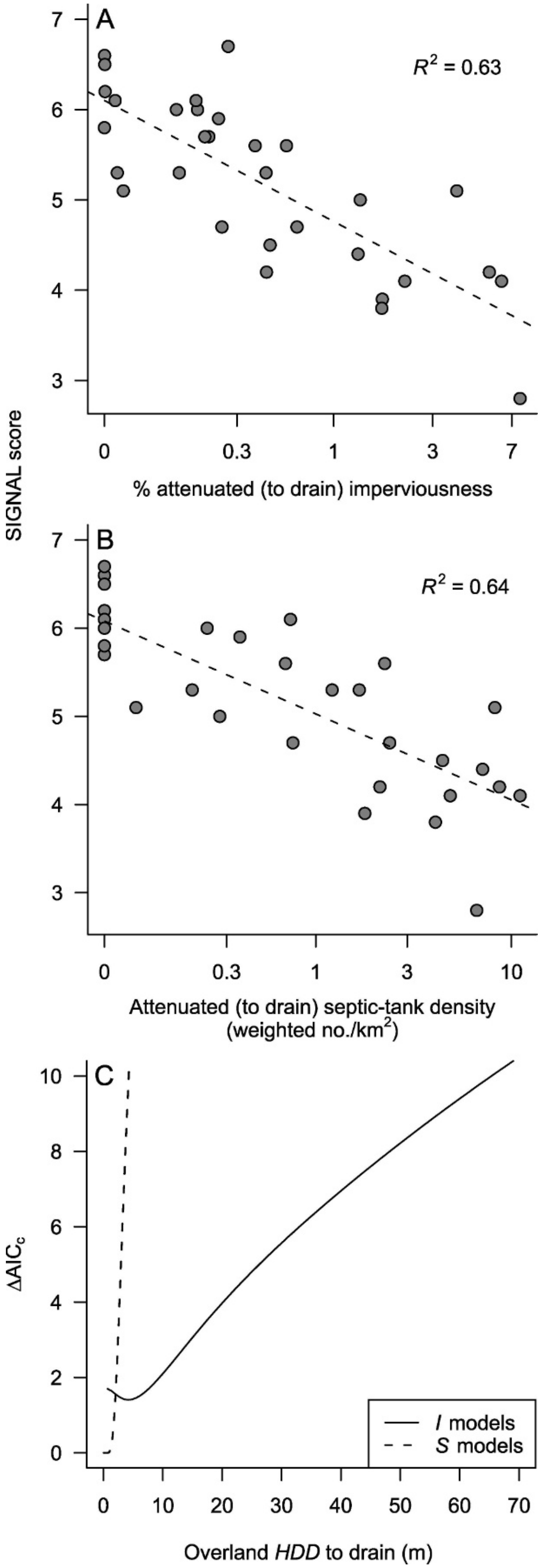

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FIG. 5. Best-fit regression models predicting the macroinvertebrate Stream Invertebrate Grade Number Average Level (SIGNAL) score in streams on the eastern fringe of the Melbourne metropolitan area. The 2 equally plausible regressors were attenuated imperviousness (*I*: the sum of impervious areas, each attenuated by an overland half-decay distance [*HDD*] of 4.3 m to the nearest stormwater drain, divided by catchment area) (A) and attenuated septic tank density (*S*: the sum of septic tanks, each attenuated by *HDD* of 0.6 m to the nearest stormwater drain, divided by catchment area) (B). A plot of difference in Akaike Information Criterion adjusted for small sample size from the best-fit model (ΔAIC_c) for *I* and *S* models against overland *HDD* to stormwater drain also is shown (C). The best-fit *HDD* is at $\Delta AIC_c = 0$; *HDD*s with $\Delta AIC_c < 2$ are equally plausible.

degradation of streams. Thus, stormwater drainage networks expand the effect of stormwater runoff from a very small proportion of impervious surfaces along the stream corridor to all connected impervious surfaces throughout a catchment.

These results suggest that great potential exists for negating stormwater effects from impervious surfaces that are not adjacent to streams through management actions that retain stormwater in catchments through evapotranspiration, infiltration, or abstraction (Walsh et al. 2005a). However, mitigation of effects from urban areas adjacent to streams is likely to be more difficult because the opportunity to intercept flows over short pathways is reduced. Thus, even if urban effects transmitted through the stormwater drainage system could be minimized through in-catchment retention, riparian buffers are likely to be essential for restoration of streams once upland urban runoff effects are mitigated. Conversely, the protective effect of riparian forests probably is reduced for instream ecosystems with upland urban areas with direct stormwater drainage connections that bypass riparian zones (Roy et al. 2005, Walsh et al. 2007).

Our results suggest that the location in a catchment of urban areas with conventional stormwater drainage networks has little effect on instream ecological condition, whereas the effects of urban areas without direct stormwater drainage connection are much more strongly dependent on their distance from and hydrologic connection to their receiving streams. Previous studies have found that urban land use closer to sampling locations has a stronger influence on streams (King et al. 2005, Van Sickle and Johnson 2008), but these studies used broad landscape classifications to define urban land use, and Van Sickle and Johnson (2008) combined urban and agricultural land use. Thus, their landuse classifications are likely to have combined several landuse effects connected to streams by different pathways with different hydraulic connectivity. Our study shows that more specific classification of urban landuse effects can reveal substantial differences in spatial dependencies between catchment and stream. Similar studies are required to test whether the patterns reported here for the eastern region of Melbourne are applicable in other locations with different climates, and catchment topography and geology.

Effects of septic tanks

Elevated N concentrations in urban and exurban streams, including streams in the eastern Melbourne region (Hatt et al. 2004), have been linked to septic-tank seepage (Steffy and Kilham 2004, Bacchus and Barile 2005, Bernhardt et al. 2008). Septic systems generally are designed to promote nitrification and convert NH_4 in effluent to mobile NO_3 (Bernhardt et al. 2008), although substantial loss of N within 10 m of septic tanks is possible (Gerritse et al. 1995a, b). Our results suggest little overland attenuation of N from septic-tank seepage in our study area. The weak relationship between NO_3/NO_2 concentrations and septic-tank density in our study (possibly caused by other N sources in our study catchments or temporal variability introduced by sampling our sites in different years) could have reduced the sensitivity of our analysis to flowpath attenuation effects. However, our analyses suggest that attenuation over short distances (less than hundreds of meters overland and less than tens of kilometers instream) is unlikely (Table 3). Further studies are required before we fully understand the relationship between instream N concentrations and septic tanks in our study area. Almost certainly, the relationship will differ and other sources will be important contributors during high flows.

In contrast to NO_3/NO_2, a mobile contaminant, concentrations of the less mobile *E. coli* were better predicted by septic-tank density when weighted by flow distance. Models of median *E. coli* concentrations (indicative of dry-weather conditions) that weighted septic-tank density by distance to stormwater drains were poorer fits than models that weighted to streams. Therefore, the main pathway from septic tanks probably is short, shallow subsurface flow along natural topographic flow paths. Other studies also suggest that *E. coli* is unlikely to be transported long distances by subsurface flows through soils (e.g., George et al. 2004).

More distant septic tanks and animal feces might influence *E. coli* concentrations more strongly during wet-weather flows. Overland flows convey fecal coliforms from fecal slurries on pastures to streams during wet weather (Tyrrel and Quinton 2003, Collins et al. 2005), but this source of contaminants might have different dynamics than subsurface leakage from septic tanks. The effects of septic tanks during high-flow events could be assessed with samples collected during high flows, but the data used in our study are inadequate for such an analysis.

Stormwater drainage was not implicated as a strong contributor to fecal pollution in our study area. However, *E. coli* concentrations are generally higher in streams in metropolitan Melbourne, which has more extensive sanitary and stormwater sewer networks and no or very few septic tanks, than in the streams we studied (MW, unpublished data). Fecal contamination from septic tanks would be a much greater problem if septic-tank leakage or overflow were connected to the stormwater system, as is the case in other parts of Melbourne (MW, unpublished data). Thus, the apparent absence of a connection between septic tanks and the stormwater drainage system inferred for our study area might not apply elsewhere.

Instream attenuation

The strength of our inferences related to instream attenuation of effects was limited by the available data. Van Sickle and Johnson (2008) concluded that urban and agricultural effects on stream fish assemblages diminish over tens of kilometers. This conclusion is consistent with our inability to infer instream attenuation of effects on macroinvertebrate assemblages in our less extensive network of streams. Finer resolution of instream distances over larger stream networks with more sampling sites could provide more robust assessment in future studies. However, the most parsimonious conclusion for management from existing evidence is that mitigation of urban effects is likely to be achieved much more effectively by using upland treatment than by relying on instream processes.

Management implications

Our results suggest that management of dry-weather fecal contamination in our study area should focus first on septic tanks within ~100 m of streams (~5% of the 20,000 septic tanks in the inventory used in our study). Management of dry-weather N contamination probably will require catchment-wide management of septic tanks, which might consist of stricter maintenance regimes or replacement with other sewage management systems.

Management of the broader ecological condition of streams in the study area (as indicated by macroinvertebrate assemblage composition) probably will require catchment-wide retention of stormwater runoff. Our results suggest that effects transmitted through stormwater drains probably are almost completely attenuated if they are directed over tens of meters of pervious upland flow path before reaching a drain. Thus, stormwater retention devices are likely to protect streams most effectively if they mimic hydrologic processes along such flow paths. Topographic flow paths permit infiltration of runoff during small-to-moderate rain events (Walsh et al. 2005a), uptake of water by terrestrial vegetation (and resultant reduction in runoff volume through evapotranspiration), retention of contaminants by soils and their biota, and slowed subsurface delivery of water to streams. These processes probably will be mimicked most effectively by dispersed in-catchment treatment measures, and the maintenance of upland drainage lines as pervious, vegetated land.

In the catchments of our study that had low attenuated imperviousness compared to total imperviousness, informal drainage such as unconnected downspouts on houses, or roads draining to forest slopes or to swales were common, and formal treatment measures were rare. Such informal drainage is likely to approximate pre-urban hydrology adequately, if the area of pervious, vegetated land downslope is large relative to the impervious areas to be drained (and if it does not lead to erosion of flowpaths or increase flood risk). However, for more extensive impervious surfaces, treatment measures that retain water and promote its loss through abstraction, such as rain-water tanks, or through evapotranspiration, such as vegetated infiltration/biofiltration systems are likely to be necessary (Walsh et al. 2005a).

More integrated research that addresses stormwater treatment technology and stream ecology is required to refine treatment designs and to develop design objectives directed at protecting stream ecosystems. Such work is required across a range of geographic regions to test the importance of catchment hydrology, topography, and geology on the effectiveness of stormwater retention and treatment on stream ecosystems.

We concur with Van Sickle and Johnson (2008) on the advantages of parametric flow-path distance-weighting models for incorporating realism and interpretability into landscape-to-stream regression models. Such models can help prioritize the scale, location, and nature of management actions for urban stream protection when applied to specific catchment-scale sources of urban effects.

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