J. N. Am. Benthol. Soc., 2005, 24(3):690–705

Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream

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Abstract. Restoration of streams degraded by urbanization has usually been attempted by enhancement of instream habitat or riparian zones. Such restoration approaches are unlikely to substantially improve instream ecological condition because they do not match the scale of the degrading process. Recent studies of urban impacts on streams in Melbourne, Australia, on water chemistry, algal biomass and assemblage composition of diatoms and invertebrates, suggested that the primary degrading process to streams in many urban areas is effective imperviousness (EI), the proportion of a catchment covered by impervious surfaces directly connected to the stream by stormwater drainage pipes. The direct connection of impervious surfaces to streams means that even small rainfall events can produce sufficient surface runoff to cause frequent disturbance through regular delivery of water and pollutants; where impervious surfaces are not directly connected to streams, small rainfall events are intercepted and infiltrated. We, therefore, identified use of alternative drainage methods, which maintain a near-natural frequency of surface runoff from the catchment, as the best approach to stream restoration in urban catchments and then used models of relationships between 14 ecological indicators and EI to determine restoration objectives. Ecological condition, as indicated by concentrations of water-quality variables, algal biomass, and several measures of diatom and macroinvertebrate assemblage composition, declined with increasing EI until a threshold was reached (EI = 0.01–0.14), beyond which no further degradation was observed. We showed, in a sample catchment, that it is possible to redesign the drainage system to reduce EI to a level at which the models predict detectable improvement in most ecological indicators. Distributed, low-impact design measures are required that intercept rainfall from small events and then facilitate its infiltration, evaporation, transpiration, or storage for later in-house use.

Key words: ecological restoration, urban, watershed, stormwater, impervious area, drainage connection, low-impact design, water-sensitive urban design, retrofit.

Degradation of stream ecosystems in urbanized catchments remains an increasing problem worldwide (Paul and Meyer 2001), despite a growing movement for urban stream restoration (Riley 1998, Carpenter et al. 2003). The objectives of stream restoration in urban areas are often presented as a return to a more natural condition, including improved water quality and biotic composition. Yet there is often tacit acceptance that many urban streams are so degraded that the probability of realizing such an objective is low (Carpenter et al. 2003).

Most attempts to restore streams in urbanized catchments have focused on reach-scale enhancement of physical habitat or reestablishment of riparian vegetation (Brown 2000). Unfortunately, ecological effects of such habitat enhancement often are not assessed (Davis et al. 2003). In almost all cases where assessments were done, changes in biotic composition were small, with only a few taxa colonizing new habitat (Larson et al. 2001, Walsh and Breen 2001, Purcell et al. 2002, Suren and McMurtrie 2005). The single study reporting significant improvements in urban stream condition following restoration (Strawberry Creek, Berkeley, California; Charbonneau and Resh 1992) involved actions beyond the channel, including removal of sewage impacts.

We propose that the poor outcomes of restoration projects involving only local-scale habitat

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enhancement result from species recruitment or persistence being constrained by catchment-scale impacts. The dominant catchment-scale impacts on biotic communities of degraded urban streams are usually associated with urban stormwater runoff (and, in many cities of the world with inadequate waste management, stormwater impacts are compounded by sewage and industrial pollution) (Walsh 2000). Attempts at restoration by instream or riparian habitat enhancement are, therefore, likely to fail because they do not match the scale of the restoration action to that of the constraining impact (Hobbs and Norton 1996, Lewis et al. 1996, Rabeni and Sowa 1996). This situation may be more strongly true for urban (cf. rural) catchments because links between the catchment and the stream are more pronounced. Conventional design of urban stormwater drainage systems connect large, dispersed areas of the catchment directly to receiving streams by pipes (Walsh 2004). Therefore, stream restoration may be more effective in urban catchments, and better matched to the dominant degrading process, if restoration was targetted to the stormwater drainage systems of the catchment rather than the stream itself.

Recent studies of small streams on an urban–rural gradient in the east of Melbourne, southeastern Australia (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Walsh et al. 2004, Newall and Walsh 2005), reported that several water-quality and biological indicators were strongly correlated with urban density, as indicated by several variables correlated with total imperviousness (TI, the proportion of a catchment covered by surfaces impermeable to water). For many indicators, much variation independent of TI and its correlates was explained by drainage connection (DC, the proportion of impervious surfaces of a catchment directly connected to streams by stormwater pipes, Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Walsh et al. 2004, Newall and Walsh 2005). All of these studies concluded that effective imperviousness (EI, the product of DC and TI, or the proportion of a catchment covered by impervious surfaces that are connected to streams by pipes) was likely to be a strong predictor of stream ecological condition. If EI is the primary cause of stream degradation, then reduction of EI, either through direct reduction of TI (by removing impervious surfaces) or reduction of DC, is likely to be an effective approach to stream restoration. Impervious surfaces are a defining element of urban land use that are difficult or impossible to remove substantially once land is developed (Beach 2001), although DC may be relatively easily reduced by retention, detention, and infiltration (Victorian Stormwater Committee 1999). However, it is unclear how much retention or infiltration is required before an impervious surface is disconnected from its receiving stream.

There were no formal stormwater treatment measures in any of the catchments of the above Melbourne studies (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005). Impervious surfaces defined as unconnected in those studies drained either to surrounding pervious surfaces, or to vegetated or earthen swales and then to streams. The primary hydrological effect of this type of indirect drainage of impervious surfaces is to intercept water from small rain events and allow infiltration or evaporation; interception efficiency would decrease in larger events. Yet, the importance of DC in explaining variation in many ecological indicators suggests that these informal interceptions may have a strong influence on the ecology of receiving streams.

Piped drainage systems allow water and associated pollutants to flow to streams more frequently than under natural conditions because surface runoff reaches streams, even during small rain events. We thus assert that the aim in achieving disconnection of hard surfaces should be complete retention of runoff from small rain events for infiltration, evapotranspiration, or reuse. Complete disconnection of an impervious surface could be achieved if the frequency of runoff from the surface is no greater than from the same parcel of land in its pre-urban condition.

We demonstrate the potential and feasibility of stream restoration through redesign of catchment drainage to reduce DC and thereby EI. We used the findings from 4 of the Melbourne studies (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005) to model relationships between EI and a range of ecological indicators, from which objectives (for EI reduction and predicted improvement in ecological indicators) can be set. Our models were not intended to confirm or test the hypothesis that EI or DC is the cause of change in each indicator, as they used the same data from which the hy-

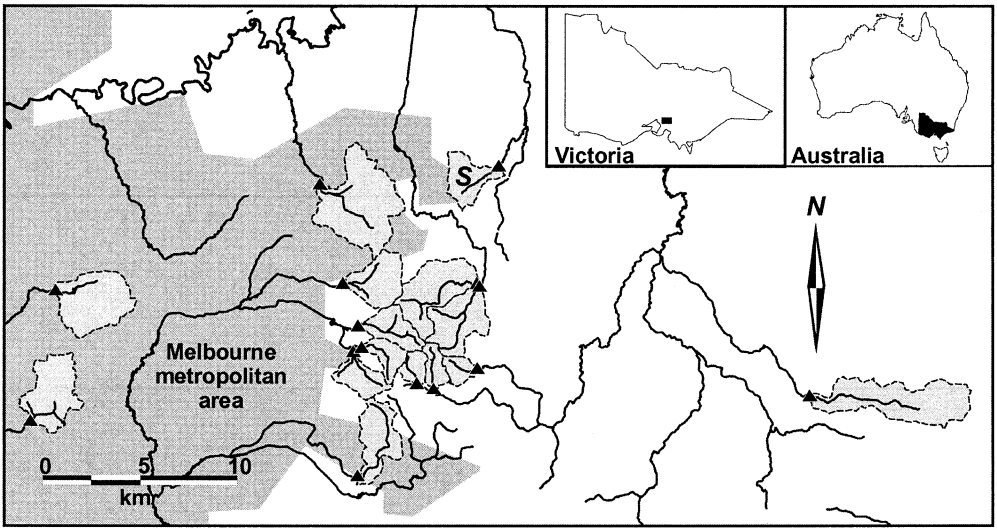

FIG. 1. Location of 15 study catchments (dashed sections with light shading) on the edge of the Melbourne metropolitan area (darker shading), southeastern Australia. The central 12 catchments span the Dandenong Ranges. S = Little Stringybark catchment. Black triangles = sampling sites.

pothesis was developed. Rather, we used models in an exploratory sense (Burnham and Anderson 2002) to set objectives for restoration. Thus, we used models to ask what reduction in EI was likely to produce a detectable change in each indicator assuming that the hypothesis of DC being the primary degrading process was true. Last, we assess the feasibility of achieving such a reduction in EI for a typical suburban development, using one of the study catchments as an example.

Methods

Study area

We used data from 4 focal studies (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005), which were conducted concurrently at the same sites on 15 first- or second-order streams with similar riparian cover, draining catchments of similar area (Fig. 1). No site was downstream of any other. The sites represented a rural-to-urban gradient and encompassed as wide a range of both imperviousness and DC as possible. We minimized variation in physiographic and climatic conditions by restricting sites to the Dandenong Ranges and surrounding hills. We ensured urban stormwater runoff was the major anthropogenic impact in the study streams by selecting only catchments with urban or forest as primary land uses, and excluding catchments with intensive agriculture. We excluded a 16th site (Monbulk Creek) used by Taylor et al. (2004), Newall and Walsh (2005), and Walsh (2004) because it was found to receive untreated graywater (i.e., nontoilet wastewater) from several properties immediately upstream. In addition, we used 1 site (Little Stringybark Creek, Fig. 1) to assess the feasibility of retrofitting a catchment's stormwater drainage system, and to predict the resulting change in a range of ecological indicators.

Catchment and instream ecological indicators

We estimated catchment area for each site using 10-m contours from the Victorian 1:25,000 topographic map series (Land Channel, Government of Victoria, Australia: http://www.land.vic.gov.au/) and local government stormwater drainage data (Whitehorse City, Monash City, Knox City, and Yarra Ranges Shire councils). We estimated TI for catchments using digital road and local government building area data together with aerial orthophotography and ground truthing across the study area. We estimated impervious surfaces connected to streams by their proximity to stormwater drains, allowing for local topography, and ground-truthed these

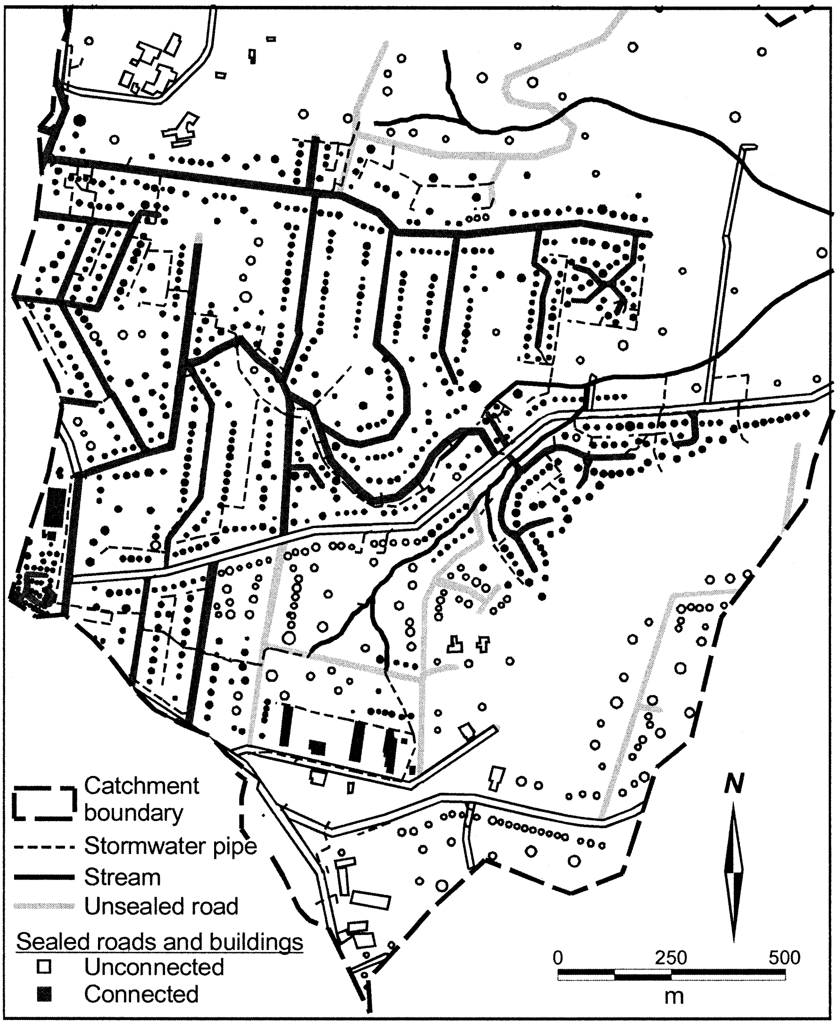

FIG. 2. Impervious surfaces (roofs and roads) and the stormwater drainage system in the upper Little Stringybark Creek catchment. Roads are classified as sealed or unsealed. Sealed roads and buildings are classified as connected or unconnected to the stream by stormwater drainage pipes.

estimates (see Taylor et al. 2004). We groundtruthed TI for Little Stringybark Creek (Fig. 2) in greater detail, following the earlier studies, by quantifying in the field the roof, garage, and driveway areas of 50 randomly selected houses. We checked drainage connection by confirming a match between supplied drainage maps and drain locations in the catchment. As a result, our TI and EI values for the Little Stringybark Creek catchment (0.086 and 0.058, respectively) differed slightly from earlier studies (0.100 and 0.055: Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005).

We considered 14 ecological indicators that were strongly correlated with the rural–urban gradient and strongly, independently correlated with DC. For water chemistry, we calculated median baseflow electrical conductivity (EC), and median baseflow concentrations of dissolved organic C (DOC) and filterable reactive P (FRP) using methods from Hatt et al. (2004). We modelled FRP using only 14 sites because we excluded an outlier that was probably subject to non-stormwater impacts (Hatt et al. 2004). For stream algae and diatoms, we estimated median chlorophyll *a* (chl-*a*) density in February, July, and November 2002 using methods from Taylor et al. (2004). We chose not to use chl-*a* data from that study for December 2001 because it was determined for only 10 of the 15 sites. We calculated the indice biologique diatomées (IBD, Lenoir and Coste 1996) using methods from Newall and Walsh (2005). For benthic macroinvertebrates, we calculated the number of families of Ephemeroptera, Plecoptera, and Trichoptera (EPT), and the biotic index SIGNAL (Chessman 1995) separately for riffle and stream-edge samples (samples collected in autumn and spring, and the data from the 2 seasons combined) as described by Walsh (2004). Determination of EPT and SIGNAL scores differed from that described by Walsh (2004) because we used only live, field-sorted animals so that derived indicators were consistent with biotic sampling protocols for the state of Victoria (EPA Victoria 2003).

Compositional similarity of diatom and macroinvertebrate assemblages in the 15 sites displayed similar patterns to univariate metrics assessed by Newall and Walsh (2005) and Walsh (2004), showing a strong correlation with urban density variables, particularly DC. Thus, we used principal curves (De'ath 1999) to permit a univariate assessment of diatom and macroinvertebrate compositional similarity (pcurve version 0.5–9, a package for R, a language and environment for statistical computing, version 1.9.1, R Foundation for Statistical Computing, Vienna, Austria; http://www.R-project.org) and reduce multivariate space to a single dimension. We calculated principal curves using $\log(x + 1)$ relative abundance diatom data of Newall and Walsh (2005) and presence–absence macroinvertebrate data for riffle samples and edge samples of Walsh (2004). For all 3 curves, we used the 1st axis of a nonmetric multidimensional scaling derived from Bray–Curtis distance matrices as a starting order for iterations (De'ath 1999).

Model selection and parameter estimation

For each ecological indicator, we assessed the fit of 4 models (described below) using the Deviance Information Criterion (DIC, Spiegelhalter et al. 2002). We used a linear regression to model a monotonic decline in ecological condition with increasing TI or EI, and a piecewise linear regression (Toms and Lesperance 2003) to model a decline in ecological condition with increasing TI or EI to a threshold beyond which no further degradation occurred. The 4 focal studies each demonstrated the importance of DC (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005), so we expected that EI would be a better predictor of all indicators than TI.

We used the WinBUGS Bayesian analysis program (version 1.4, the BUGS project; http://www. mrc-bsu. cam. ac. uk/ bugs/ winbugs/ contents.shtml) to estimate the joint posterior probability distributions of model parameters with the data for each indicator (Spiegelhalter et al. 2003). Bayesian inference is a statistical tool increasingly being used by ecologists, and is advantageous because it includes prior knowledge in quantitative models and produces posterior probability distributions that provide a direct measure of belief about models or parameter estimates (Ellison 2004). For all models,

$$Y_i \sim \text{Normal}(\mu_i, \sigma_i), \qquad [1]$$

where Y_i was the observed indicator value at stream i, assumed to be a sample from a normally distributed population with a true population mean μ and standard deviation σ. The prior distribution used for σ-values was an uninformative gamma distribution (mean and precision both = 0.001). For all indicators, linear models were of the form:

$$\mu_i = \alpha + \beta I_i \qquad [2]$$

where I_i is the imperviousness (EI or TI) for stream i. Prior probability distributions for α and β were uninformative normal distributions with mean = 0 and precision = 10^{-6}. For chl-*a* and for water-quality variables, the piecewise model was of the form of an increase with a slope β from a minimum, α, at $I = 0$, to a maximum γ, thus:

$$\mu_i = \max\begin{pmatrix}\alpha + \beta I_i \\ \gamma\end{pmatrix}. \qquad [3]$$

For all other variables, the piecewise model was of the form of a decrease with a slope β from a maximum, α, at $I = 0$, to a minimum γ, thus:

$$\mu_i = \min\begin{pmatrix}\alpha - \beta I_i \\ \gamma\end{pmatrix}. \qquad [4]$$

For piecewise models, the prior distribution for β was a uniform distribution from -0.1 to 1000, and the priors for α and γ were uniform distributions within the feasible range for each indicator. Models were run with 3 Markov chains, a burn-in of 5000 iterations, followed by 5000 iterations thinned every 15 (see Spiegelhalter et al. 2003).

For piecewise models using EI, we also estimated the minimum threshold level of EI at which each indicator was predicted to = γ (i.e., the minimum EI at which maximum degradation is predicted). We also estimated posterior distributions of each indicator for 4 levels of EI: the current EI of Little Stringybark catchment (EI = 0.055), the 2 values of EI achievable if 12 and 15 ha of impervious surfaces in the Little Stringybark catchment were disconnected (EI = 0.028 and 0.020, respectively), and the reference condition (EI = 0).

Feasibility assessment

To assess the degree of retention required to achieve disconnection of impervious surfaces in the Little Stringybark catchment, we first estimated the frequency of surface runoff from a parcel of naturally forested land of the same size as a typical residential allotment (600 m^2) not crossed by a drainage line (using the rainfall–runoff model of Chiew and McMahon 1999). We then estimated the minimum mean size of a rain event (in mm/d) required to produce this surface runoff (using the 1965–1975 rainfall record for station 86234 Croydon, mean annual rainfall = 937 mm; Australian Bureau of Meteorology, Melbourne). We considered an impervious surface disconnected (i.e., noneffective) if all stormwater runoff from the surface was retained for infiltration, evapotranspiration, or reuse from rain events up to this size.

We estimated the frequency of surface runoff from a connected impervious surface to compare the frequency of runoff between effective and noneffective impervious surfaces. We estimated runoff to be generated from an impervious surface by rainfall events of >1 mm/d.

We tested the feasibility of reducing EI of a typical residential allotment of 600 m^2 with TI of 0.5 (200 m^2 of roof, 100 m^2 of pavement) to 0, using the MUSIC model (Cooperative Research Centre for Catchment Hydrology 2003) to simulate the runoff frequency with low-impact design (LID) technologies applied at the allotment scale only. We estimated water demand from rainwater tanks after Coombes et al. (2003), allowing overflow frequencies to be simulated.

We also investigated the feasibility of disconnecting 15 ha of impervious surfaces in the Little Stringybark catchment without reliance on treatment within private properties. We gathered information on the drainage system, roads, roofs, and open space in the catchment to determine the opportunities for measures that could reduce the runoff frequency to predevelopment levels (Fig. 2).

We considered the following options in creation of retrofit scenarios: rainwater tanks, pervious pavements, infiltration wells, infiltration trenches, buffer strips, grass swales, rain-gardens, infiltration basins, detention basins, ponds, bioretention systems, and constructed wetlands (Victorian Stormwater Committee 1999). Selection of appropriate techniques was based on criteria that considered site conditions including slope, landslip hazard, soil infiltration capacity, and depth to groundwater. We also obtained the cadastre (outlining individual property parcels), and mapped pavement quality (to assess the feasibility of retrofitting with porous pavements), available space along roads (for swales and bioretention systems), and other open space (for wetlands and infiltration basins). We developed alternative retrofit scenarios and compared their performance against our criterion of disconnection using MUSIC (Cooperative Research Centre for Catchment Hydrology 2003).

Results

Relationships between ecological indicators and EI

All ecological indicators assessed showed strong degradation with increasing EI to a threshold of maximum degradation (Fig. 3). The principal curve for diatom assemblages explained 34% of variance in species composition, whereas the principal curves for macroinvertebrates within riffles and edges explained 46 and 31% of the variance in family composition, respectively.

For all but 3 indicators, the best-fit model was the piecewise regression against EI (Table 1, Figs 3 and 4). For DOC, a linear regression with TI was the best fit (Table 1, Fig. 3B) and for chl-*a* in February and July, a linear regression with EI was the best fit (Table 1, Fig. 3D and E, re-

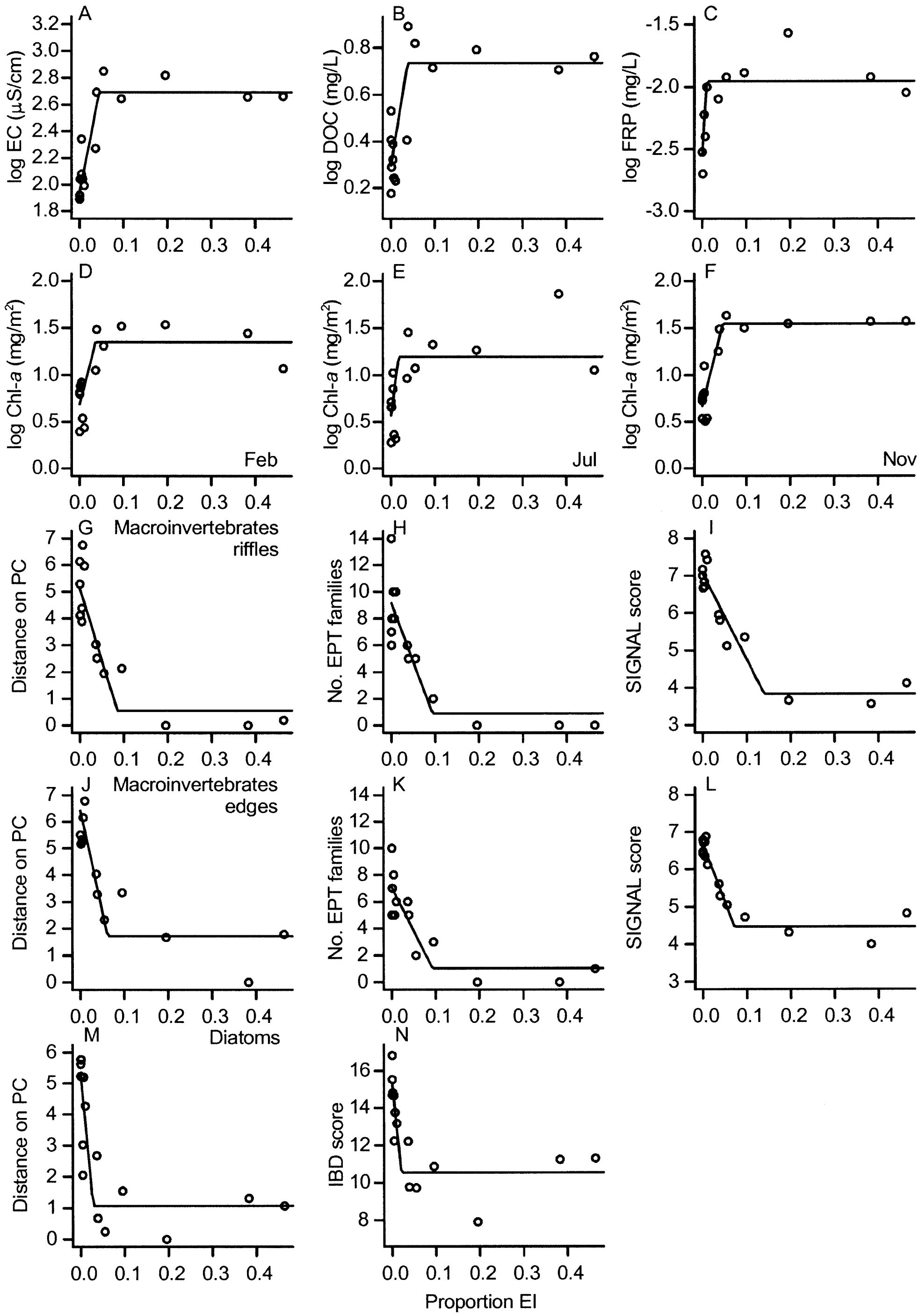

FIG. 3. Relationships of each of the variables listed in Table 1 with effective imperviousness (EI) in 15 sites in the east of Melbourne (14 for filterable reactive P [FRP], with outlier excluded). The line of best fit is a piecewise regression modelling degradation with increasing EI to a threshold beyond which no further degradation is predicted using median parameter estimates. EC = electrical conductivity, DOC = dissolved organic C, chl-*a* = chlorophyll *a* density, PC = principal curve, EPT = Ephemeroptera, Plecoptera, and Trichoptera, IBD = indice biologique diatomée.

TABLE 1. Deviance Information Criteria (DIC) for 4 models of each of 3 measures of median baseflow water quality (as electrical conductivity [EC], dissolved organic C [DOC], and filterable reactive P [FRP]); 3 estimates of median benthic chlorophyll *a* density (chl-*a*) in February, July, and November 2002; 2 indices based on diatom assemblage composition (as distance on a principal curve [PC]) and indice biologique diatomée [IBD]; and 3 metrics based on macroinvertebrate assemblage composition (as PC, number of families of Ephemeroptera, Plecoptera and Trichoptera [EPT], and SIGNAL score) measured for riffles and edges. EC, DOC, FRP, and chl-*a* data were log-transformed before analysis. The best model as indicated by the minimum DIC for each variable is shown in bold.

	Median baseflow water quality			Median chl-*a* density			Diatoms		Macroinvertebrates					
									Riffles			Edges		
Model	EC	DOC	FRP	Feb	Jul	Dec	PC	IBD	PC	EPT	SIGNAL	PC	EPT	SIGNAL
Piecewise effective imperviousness	**2.6**	2.2	**−3.7**	22.9	24.3	**11.1**	**54.3**	**63.0**	**56.6**	**71.9**	**26.3**	**52.9**	**69.2**	**12.0**
Piecewise total imperviousness	25.7	5.0	16.4	19.2	23.1	37.5	71.3	71.4	76.7	76.3	36.2	60.7	69.4	34.2
Linear effective imperviousness	13.5	2.0	8.9	**16.5**	**19.3**	15.6	66.9	66.9	59.1	78.5	39.9	78.5	69.6	36.3
Linear total inperviousness	11.6	**1.6**	9.2	19.0	19.5	17.6	66.8	66.8	60.6	76.9	44.2	76.9	69.6	36.1

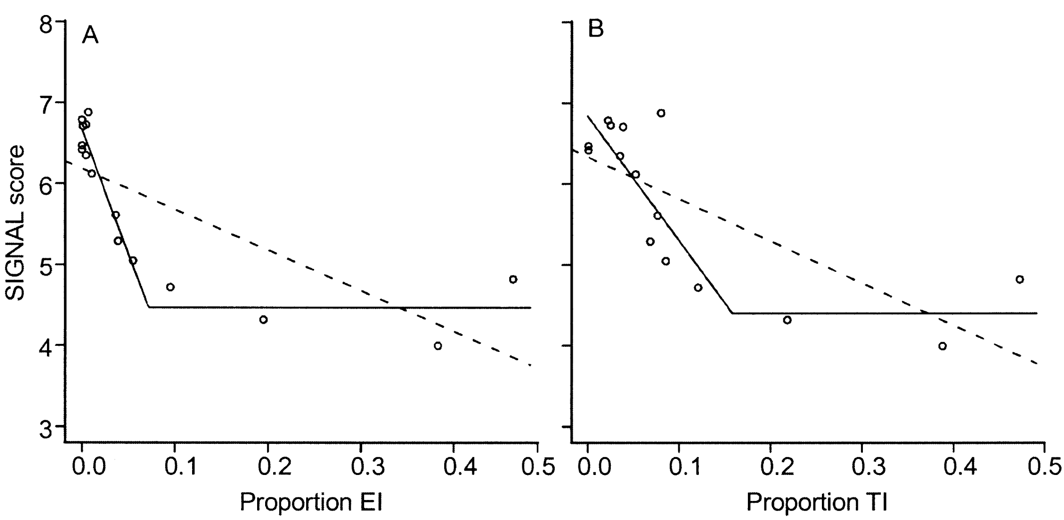

FIG. 4. SIGNAL scores for macroinvertebrates in edge habitats plotted against effective imperviousness (EI) (A) and total imperviousness (TI) (B) as an illustration of the 4 models assessed for goodness of fit (Table 1). Solid lines are piecewise regressions and dashed lines are linear regressions.

TABLE 2. Medians and 90% credible intervals of parameters for piecewise regression models of the 14 ecological indicators in relation to effective imperviousness (EI). The parameters α, β, and γ are those in equations 3 and 4. The threshold parameter is the value of EI at which the break in the regression is reached. Prior distributions for α and γ were uniform within the range indicated for each variable. Definitions for abbreviations as in Table 1.

	α	β	γ	Prior range for α and γ	Threshold
Median baseflow water quality					
$\log_{10}$ EC (μS/cm)	1.9 (1.8, 2.1)	17 (10, 39)	2.7 (2.6, 2.8)	1–4	0.05 (0.02, 0.07)
$\log_{10}$ DOC (mg/L)	0.3 (0.2, 1.3)	12 (5, 697)	0.7 (0.5, 0.9)	0–2	0.04 (0, 0.08)
$\log_{10}$ FRP (mg/L)	−2.6 (−2.8, −2.3)	57 (13, 490)	−2 (−2.1, −1.8)	−3–0	0.01 (0, 0.04)
$\log_{10}$ chl-*a* density (mg/m²)					
Feb	0.7 (0.5, 1.3)	19 (7, 779)	1.4 (1.0, 1.6)	0–2	0.04 (0, 0.09)
Jul	0.6 (0.3, 1.5)	37 (7, 864)	1.2 (0.8, 1.6)	0–2	0.02 (0, 0.11)
Nov	0.7 (0.5, 0.8)	19 (12, 44)	1.5 (1.4, 1.7)	0–2	0.05 (0.02, 0.07)
Diatoms					
PC	5.3 (4.4, 6.2)	−155 (−740, −70)	1.1 (0.3, 2.0)	0–10	0.03 (0.01, 0.06)
IBD	15.3 (14.3, 16.4)	−231 (−612, −88)	10.5 (9.5, 11.5)	1–20	0.02 (0.01, 0.05)
Macroinvertebrates: riffles					
PC	5.1 (4.3, 5.9)	−54 (−96, −25)	0.6 (0, 1.6)	0–10	0.08 (0.04, 0.17)
EPT	9.2 (7.8, 10.7)	−88 (−196, −46)	0.9 (0.1, 2.8)	0–20	0.09 (0.04, 0.16)
SIGNAL	7.1 (6.7, 7.4)	−23 (−44, −15)	3.8 (3.3, 4.5)	1–10	0.14 (0.07, 0.22)
Macroinvertebrates: edges					
PC	6.4 (5.6, 7.3)	−75 (−177, −34)	1.7 (0.6, 2.9)	0–10	0.06 (0.02, 0.14)
EPT	7.1 (6, 8.3)	−65 (−260, −29)	1.0 (0.1, 2.9)	0–20	0.09 (0.02, 0.20)
SIGNAL	6.7 (6.5, 6.9)	−31 (−40, −21)	4.5 (4.2, 4.7)	1–10	0.07 (0.06, 0.10)

spectively). These 3 exceptions were characterized by variable, low values for all streams at near-zero EI and variable, high values for streams at higher EI. Differences in DICs among the 4 models for these 3 indicators were small (Table 1), so we used piecewise regressions against EI for subsequent analyses on all indicators to have a consistent theoretical basis to the models.

Threshold EIs for the water-quality indicators, chl-*a* density, and diatom assemblage indicators were generally low (medians 0.01–0.05, Table 2; Fig. 3M and N), whereas threshold EI values for macroinvertebrate assemblage indicators were generally higher (medians 0.06–0.14, Table 2; Fig. 3G–L). Uncertainty of parameter estimates (as indicated by the credible limits of the threshold estimate) varied among indicators (Table 2). Uncertainty was high for the 3 indicators where the EI piecewise regression was not the best-fit model (i.e., chl-*a* in February and July and DOC) and for edge EPT. Uncertainty was low for EC, chl-*a* in November, IBD score, the principal curve for diatom assemblages, and edge SIGNAL score (Table 2).

For EPT and SIGNAL score, for which legislated biological objectives for ecosystem protection exist, Little Stringybark failed to meet the biological objectives (Fig. 5). For all models, the predicted value for EI = 0 showed little or no overlap with the distribution of estimates for EI = 0.055 (current Little Stringybark condition, Fig. 5). Disconnection of 12 and 15 ha of impervious surfaces from this catchment was predicted to move median estimates of most indicators closer to the reference condition than the Little Stringybark condition, although disconnection of 15 ha was required to result in overlap of <0.05 of the distribution for almost all indicators (Fig. 5). Even with this level of disconnection, little change in FRP was predicted (Fig. 5C). The models predicted a high probability of

Little Stringybark Creek meeting the Victorian biological objectives for SIGNAL score under either disconnection scenario, and a low probability of meeting the EPT objective. However, distribution of EPT richness among sites with near-zero EI (Fig. 5H, K) suggested that the EPT objectives would not be met for several reference sites in the data set, even though they experienced little human impact.

Feasibility assessment

Surface runoff was estimated to occur from a forested parcel in the Little Stringybark catchment on ~4% of days (mean = 15 d/y). Such runoff would be generated by rain events >15 mm/d, which in eastern Melbourne occurs mostly between August and October. In contrast, surface runoff from connected impervious surfaces was estimated to occur on ~33% of days (mean = 120 d/y), and was spread more evenly over the year.

Many design approaches resulted in surface runoff from the 600 m^2 allotment in the Little Stringybark catchment on no more than 4% of days, thereby reducing EI from 0.5 to 0. As an example, one successful design incorporated a series of treatments comprising a rainwater tank, porous pavement, and a rain garden. The roof area (200 m^2) was diverted into a 6 kL rainwater tank, used for internal nonpotable use (150 L/d) and garden watering (200 kL/y, with timing of demand matched to potential evapotranspiration). All paved areas (100 m^2) used porous paving with an infiltration capacity of 2 mm/h. Overflow from both the roof area and porous pavement were directed into a rain-garden (15 m^2), with an underlying infiltration capacity of 2 mm/h, and a maximum ponding depth of 300 mm. Overflow from the rain-garden was conveyed via a pipe to the conventional stormwater system (i.e., no estate-scale or downstream treatments were assumed).

Scenarios of drainage redesign at the catchment scale relying on end-of-pipe wetlands and detention ponds failed to maintain the predevelopment runoff frequency from 15 ha of impervious surfaces. This criterion was achieved in several scenarios using more distributed, at-source approaches. Only dispersed treatments at the allotment- or street-scale detained the higher-than-natural runoff depth from impervious areas, for infiltration, evapotranspiration, or reuse.

Discussion

A conceptual framework of urban stormwater impacts on streams

We used evidence of the likely importance of EI as a degrader of streams in urban catchments to propose a new catchment-based approach to stream restoration. We argue that the most likely dominant process degrading stream communities can be reversed by preventing increased frequency of surface runoff generated by EI during small to moderate storms (Table 3).

If multiple impervious surfaces were connected to the stream by stormwater pipes (as is the traditional approach to stormwater management), then runoff from these more frequent, smaller storms of <15 mm would be delivered to the stream as frequent disturbance events. Such disturbances would result from interactions among physical disturbance and chemical stressors (Table 3), and could explain the observed changes in macroinvertebrate assemblage composition in eastern Melbourne streams (Walsh 2004, Newall and Walsh 2005), whereas frequent pulses of high-nutrient water could explain observed increased biomass of benthic algae (Taylor et al. 2004).

A secondary effect of increased runoff from impervious surfaces delivered to the stream by pipes is the reduction of infiltration and consequent lowering of the water table and base flow. Leaky water supply and sewerage infrastructure or urban irrigation may increase urban water tables in some areas (Nilsson et al. 2003), but such effects are likely minor in the eastern fringe suburbs of Melbourne. Lower water tables tend to reduce base flow and increase instream retention time of high-nutrient storm inputs. In turn, increased retention time, in concert with increased algal growth, can lead to nocturnal O_2 depletion. The direct connection of impervious surfaces to streams also increases the risk of toxicity occurring from spills in the catchment between rain events (Table 3).

If dispersed, small-scale stormwater treatment was applied to intercept rainfall of ≤15 mm in the urbanized catchment, then instream impacts resulting from stormwater runoff would be restricted to larger rain events (Table

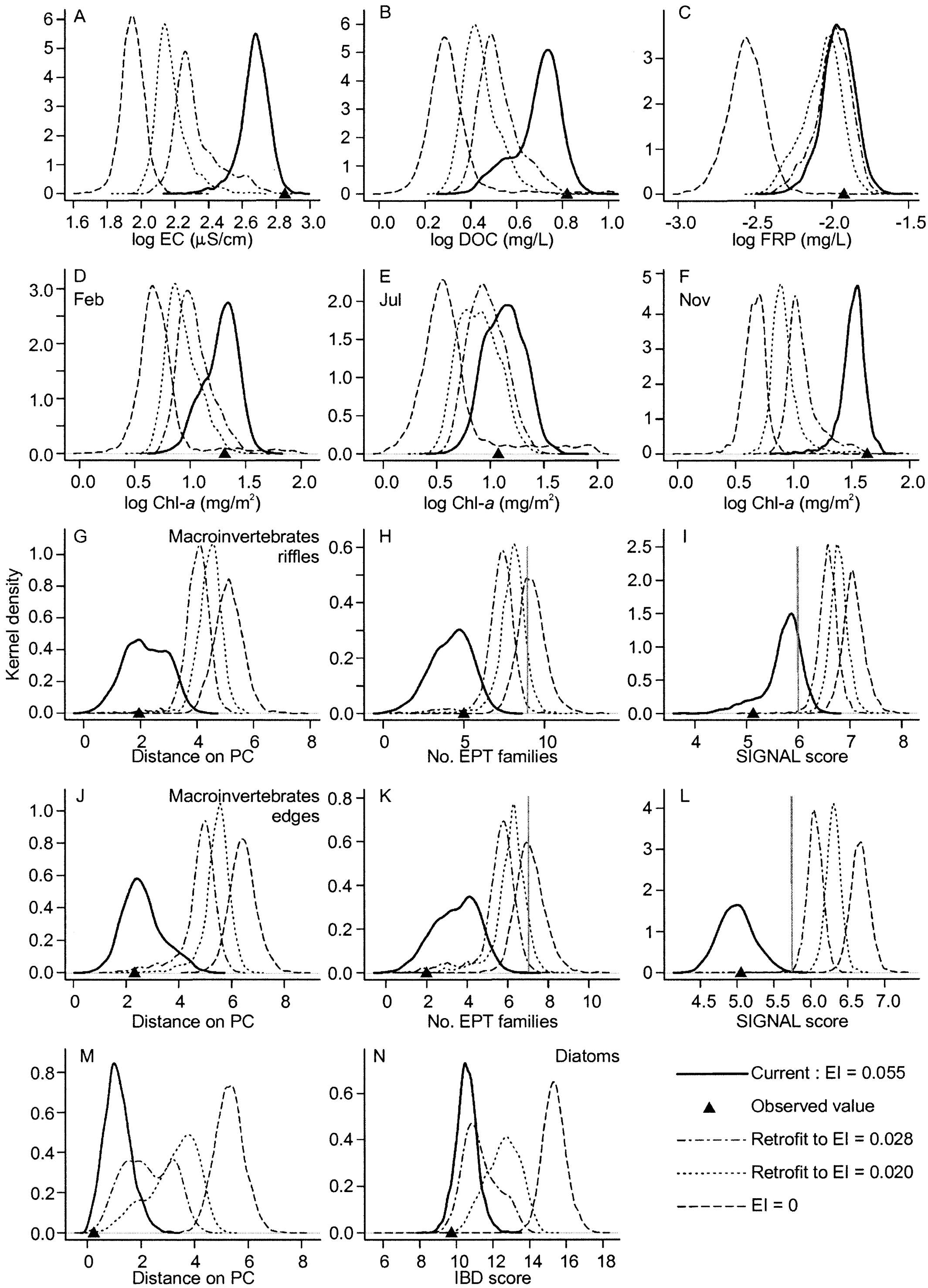

FIG. 5. Posterior distributions (as kernel densities) of variables listed in Table 1 from piecewise regression models for 4 values of effective imperviousness (EI): the current EI at Stringybark Creek (0.055), the value that would be achieved through retrofitting 12 ha and 15 ha of impervious surfaces (0.028 and 0.020, respectively),

TABLE 3. Conceptual framework of stormwater impacts to stream ecosystems, comparing 2 urban retrofit scenarios and the pre-urban condition. Scenarios are based on a hypothetical stream in the Dandenong Ranges (rainfall frequencies based on 1965–1975 data for Croydon, Victoria; Australian Bureau of Meteorology, Melbourne, Victoria), with the 2 urban scenarios assuming a total imperviousness >10%. TSS = total suspended solids.

<table>
<tr><th>Storm size and frequency</th><th>Conventional urban design[a]</th><th>Low-impact design[b]</th><th>Pre-urban land</th></tr>
<tr><td>No effective rainfall (<1 mm/d: ~67% of days)</td><td>Low water table, low base flow; high P and N concentrations; variable, mostly low turbidity; high pollutant spill risk; high algal biomass, variable O_2; low invertebrate and fish diversity</td><td colspan="2">Plentiful baseflow of high-quality water fed by subsurface flows; good quality habitat supporting diverse biota</td></tr>
<tr><td>Small–moderate rain events (1–15 mm/d: ~29% of days)</td><td>Moderate to large discharge increase, possible substratum movement, and bank erosion; inflow with high N, P, TSS, and toxicant concentrations; loss of sensitive biota (flow disturbance–toxicant interactions); filamentous algal and diatom growth stimulated</td><td colspan="2">No surface runoff; replenished subsurface-fed baseflow; negligible physical disturbance from slightly higher flow</td></tr>
<tr><td>Large rain events (>15 mm/d: ~4% of days, mostly in wet season)</td><td>Large flood; major incision and bank erosion; large inflow of N, P, TSS, and toxicants; loss of all sensitive biota; smothering/scouring of algae</td><td>Large flood; substratum movement and bank erosion; inflow with high N, P, TSS, and toxicant concentrations; loss of sensitive biota, but species adapted to annual flooding likely to recolonize</td><td>High discharge; substratum movement; increased N, P, and TSS concentrations; temporary loss of some species, but those adapted to annual flooding will recolonize</td></tr>
</table>

[a] All impervious surfaces drained by pipes or sealed drains directly to stream
[b] Runoff from impervious surfaces retained up to a 15-mm rain event

3). Increased catchment TI is likely to cause higher, more intense flows from larger rain events than those in the pre-urban state (and also probably higher pollutant concentrations), even with the provision of dispersed stormwater retention and treatment measures. However, timing of storm events would be consistent with the pre-urban stream, primarily occurring during the wettest part of the year. Thus, ecological impacts of these larger events may be minor compared to the impact of frequent disturbances from small storms in traditionally drained urban catchments because they are closer to the frequency and seasonal timing of disturbance to which lotic biota are adapted (Resh et al. 1988).

Key processes for stream restoration in urban catchments

Our proposed approach to stream restoration in urban catchments is consistent with the first 4 key processes identified by Hobbs and Norton (1996) as essential for successful integration of ecological restoration into management. Asso-

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and EI = 0 (being equivalent to the posterior distribution of the parameter α). Solid triangle in each panel indicates the value of the variable observed at Stringybark Creek. Vertical lines in plots for EPT and SIGNAL indicate the biological objectives for the protection of rivers and streams in Victoria (EPA Victoria 2003). Definitions for abbreviations as in Fig. 3.

ciated with their 1st process, we identified impervious surfaces directly connected to streams by stormwater pipes (Hatt et al. 2004, Taylor et al. 2004, Walsh 2004, Newall and Walsh 2005) as the dominant cause of degradation in streams of urban catchments.

Associated with Hobbs and Norton's (1996) 2nd process, we identified the most important elements of alternative drainage design useful in reversing degradation. We proposed that maintenance of a near-natural frequency of surface runoff should be the critical objective of stormwater management. Impervious surfaces for which this objective is achieved can be classified as unconnected and, thus, should have minimal impact on receiving streams. This approach is consistent with stormwater management priorities aimed at maintaining the catchment water balance at pre-urban levels (Tourbier 1994, Stephens et al. 2002). However, it differs markedly from commonly applied objectives for stream protection, such as maximum limits to TI or minimum limits to forest cover (e.g., Stephens et al. 2002). We hypothesize that TI can potentially be maintained as long as EI is reduced.

Associated with Hobbs and Norton's (1996) 3rd process, we determined a range of possible goals for reestablishing ecological condition in a sample stream by modelling relationships between a range of ecological indicators and EI. By testing several scenarios, we identified the amount of impervious surfaces requiring disconnection (i.e., redesigned to allow interception of rain events of $\leq$15 mm), to produce a detectable improvement in condition as predicted by the models.

Associated with Hobbs and Norton's (1996) 4th process, the range of indicators we assessed provide easily observable measures of success, although a fuller assessment of the stream ecosystem is achievable by including measures of ecological processes (Bunn et al. 1999, Grimm et al. 2005, Groffman et al. 2005, Meyer et al. 2005).

Hobbs and Norton's (1996) 5th key process involved development of practical techniques for implementing restoration goals. Many techniques exist for achieving dispersed retention of stormwater for infiltration and treatment (e.g., Victorian Stormwater Committee 1999). However, in designing a catchment drainage system, a balance is required between the extent of reuse, evapotranspiration, and infiltration. Reuse, as in the residential allotment retrofit we described (where 150 L/d were exported from the catchment by sanitary sewers) may, if applied extensively, reduce base flows to receiving streams. However, such losses may have no effect on catchment water balance if these losses are counteracted by reduced evapotranspiration from forest loss, or water import into the catchment by a reticulated water supply system.

Techniques promoting water infiltration within the catchment should be implemented so that they maintain stream base flows through subsurface flows. However, the choice of infiltration method should protect values associated with aquifers and nearby buildings or roads and avoid risks associated with slope instability. Biofiltration systems (Lloyd et al. 2002), in particular, have great potential to avoid such problems, while mimicking shallow subsurface flows and efficiently treating pollutants such as NO_3-N that can be transported through subsurface flows. To achieve a near-natural water balance in the catchment, application of LID techniques will likely involve selecting appropriate tools at a mix of scales—allotment, streetscape, precinct and regional—to achieve the required target.

Final thoughts

The correlational data in our paper cannot provide strong evidence of causal factors driving stream degradation in urban catchments. However, the prevalence of DC as a key factor explaining a range of ecological indicators suggests a strong potential for ecological restoration of streams through catchment drainage retrofit. The models of relationships between EI and a range of ecological indicators provide strong, quantitative hypotheses testable by experimental manipulation of catchment drainage. DC (and therefore EI) is a manageable attribute of urban land, so it is all the more important that the potential of manipulating EI as a tool for stream restoration be robustly assessed.

The strong threshold relationship observed in all ecological indicators suggests that, for streams in highly impervious catchments and with high EI, a large disconnection effort would be required before instream ecological change would be predicted. However, our assessment of management options at the allotment scale suggest that the target level of EI = 0.02 may be possible up to TI = 0.50. Reducing EI to very low levels is likely to become increasingly difficult as TI increases >0.50. Nevertheless, these limitations suggest that stream restoration through catchment retrofit in most exurban and many suburban areas in cities of the developed world (which typically have TI ≤ 0.50; CWP 2003) is possible.

In conclusion, we argue that catchment-scale stormwater drainage is the constraining factor for stream restoration in urbanized catchments. We are not disputing that increasing instream habitat complexity and riparian vegetation are important elements of restoration. However, the effectiveness of such local-scale measures is likely to be limited if EI-related impacts constrain stream communities. The importance of riparian processes to stream ecological condition, in particular, may be compromised by the catchment-scale effects of conventional stormwater drainage (Groffman et al. 2003, Taylor et al. 2004, Walsh 2004). Conversely, recovery of some ecological indicators such as algal biomass may be constrained if stormwater impacts are removed without reestablishing riparian vegetation. The relative efficacy of stream restoration by drainage retrofit and by local-scale habitat restoration requires controlled experiments to better understand interactions across scales.

Acknowledgements

We thank Belinda Hatt, Sally Taylor, Simon Roberts, and Peter Newall for contributions to the earlier studies forming the basis of this paper, Alex Leonard for discussions on the conceptual framework, and Pascal Horton for assistance with developing scenarios to test feasibility of retrofitting the Little Stringybark catchment. This work was part of the CRC for Freshwater Ecology projects D210 and B270, partly funded by Melbourne Water. The manuscript was greatly improved by the suggestions of Jack Feminella and 2 anonymous referees.

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Received: 2 March 2005
Accepted: 1 July 2005