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Spatial analysis of anthropogenic river disturbance at regional and continental scales: identifying the wild rivers of Australia

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Abstract

A method for assessing anthropogenic river disturbance is described. The grid-based spatial modeling procedure computes indices of disturbance for individual stream sections. These indices rank streams along a continuum from near-pristine to severely disturbed. The method couples geographical data, recording the extent and intensity of human activities known to impact on river condition, with a Digital Elevation Model (DEM) used for drainage analysis. It was developed to produce the first nation-wide assessment of river disturbance from which Australia's least disturbed or 'wild' rivers were identified. A national summary of the extent and the potential impact of human activities is presented, calculated from the disturbance index values computed for more than 1.5×10^6 stream sections with a total length of over 3×10^6 km. Index values close to the undisturbed end of the continuum are rare, especially among large rivers. Most of the least disturbed streams are predicted to lie within the monsoonal tropical north or the arid/semi-arid center of the continent.

The disturbance indices generated provide a comprehensive and consistent characterization of river and catchment disturbance that has applications beyond the identification of wild rivers. These include identification of priorities for rehabilitation and restoration; development of systematic survey strategies for aquatic, riparian and estuarine biota and identification of reserve networks for river systems. However, these applications depend on validating the correlation between river disturbance indices and intensively sampled physical and biological indicators of river condition. © 2002 Elsevier Science B.V. All rights reserved.

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1. Introduction

Human activities have had a profound impact on river systems throughout the world. River regulation has dramatically altered seasonal flow patterns, and reduced connectivity between rivers and their floodplains (Petts, 1984; Kingsford, 2000). The impacts of catchment land uses include both changes to water quality from point and diffuse sources of pollutants and alteration to flow regimes, for example, as a result of increased run-off following the clearing of native vegetation (Boulton and Brock, 1999; Pen, 1999). These impacts are cumulative and often synergistic (Lake et al., 2000) and have serious consequences for aquatic fauna, including the loss of habitat (Jackson, 1997), fragmentation and isolation of populations (Vaughn and Taylor, 1999) and direct mortality (Lytle and Peckarsky, 2001).

A very large number of indicators of river health or condition have been developed to assess the impacts of disturbance (Boulton, 1999). These indicators include aspects of the physical or chemical characteristics

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of the stream (Jolly et al., 1996; Liston and Maher, 1997; Brierley et al., 1999; Hart et al., 1999; Maddock, 1999; Maher et al., 1999; Doyle et al., 2000) or less commonly, integrated ecological, hydrological and geomorphological variables (Ladson et al., 1999). Much of the recent focus has been on biological indicators, from single taxa used to monitor specific pollutants (Bargagli et al., 2002) to complex indicators of community composition (Chessman et al., 1999; Harris and Silveira, 1999; Smith et al., 1999).

All of these indicators are site-based, but sites may not be representative of larger units, such as long sections or entire rivers, for example, because of localized variability in community composition (Downes et al., 2000). Yet landscape level or regional assessment is needed for policy development, planning and management. This is the scale at which the cumulative effects of impacts are evident (Boulton, 1999). Ethical stewardship of the environment requires that society monitor and assess environmental changes at the national scale (O'Neill et al., 1997).

In this paper, we describe a method to spatially analyze anthropogenic disturbance of streams at regional and continental scales. The method provides a consistent characterization of entire river networks that can complement site based indicators of river condition. It is suitable for application in data-sparse regions, countries or continents, requiring only data commonly available from basic topographic and land use mapping.

The method was developed as a tool to assist the identification of Australia's remaining 'wild' rivers (Stein et al., 2001). Wild Rivers are those rivers for which "the biological, hydrological and geomorphological processes associated with river flow have not been significantly altered by modern or colonial society", a definition adopted by the Australian Heritage Commission from a commissioned report by BIOSIS Research in 1993. This report also identified a key set of flow regime and catchment indicators condensed from an extensive literature review to be used in conjuction with set criteria to determine the wild river status.

Rather than the proposed binary concept, i.e. wild/not wild based upon arbitrary thresholds of indicator values, our method is based on a continuum concept with disturbance ranked on the continuum from near-pristine to highly degraded. It does not require that individual physical processes be modeled to provide accurate measures of pollutant or sediment loadings or alterations to the flow regime. Rather, it uses surrogate measures to link the impact of human activities with the stream via overland flow within the catchment and via stream topology to derive empirical indices of river disturbance. The method: (1) considers both point and diffuse impacts on water quality; (2) includes both catchment and in-stream factors that alter flow regimes; and (3) can be applied at a very wide range of scales over large and diverse areas with limited data. The indices are spatially modeled using a grid-based procedure implemented in the ArcInfo Geographic Information System (GIS) software (ESRI, 1996). A Graphical User Interface (GUI) allows easy selection of options for computation, display and query of disturbance indices.

Here, we demonstrate its application to the first national assessment of anthropogenic disturbance for Australian streams, as the initial step in identifying the remaining wild rivers. Wild rivers are identified within the context provided by an assessment of all rivers.

2. Methods

Our method is based on the assumption that (a) the intensity and extent of human activities within the catchment and (b) in-stream structures that alter the flow regime, provide surrogate indicators of the extent of disturbance of natural river processes. The degree to which the hydrological, geomorphological and biological processes of a stream have been altered can be considered to form a continuum from severely degraded to near-pristine or 'wild'. Accordingly, a continuous measurement scale derived from primary attribute data provides greatest flexibility in classification for a wide range of potential applications. The continuous rating scale for disturbance also allows user specifications of quantitative thresholds for the definition of wild rivers.

The procedure distinguishes direct changes to the flow regime, e.g. as caused by a dam, from indirect impacts on the hydrological, geomorphological or ecological functions of the stream resulting from anthropogenic changes in the catchment. An index approach was adopted because it was more easily understood by, and communicated to a wide range of potential users and because it reflected the qualitative nature of much of the primary data. The indicators chosen were those identified as key catchment or flow regime indicators by a wide range of studies. A rating is derived for every stream section or link, i.e. the portion of a stream between junctions, based initially only on disturbances to that section or within its immediate sub-catchment. For each stream section, a contributing catchment area is determined from a drainage analysis based on a "digital elevation model" (DEM).

Factor scores are computed for each of four major sources of catchment disturbance with potential to significantly alter river processes: (1) land use activity; (2) settlements and structures; (3) infrastructure; and (4) extractive industries and other point sources of pollution. Scores reflect both the spatial extent and potential magnitude of impact from the disturbance and, in the case of point sources, proximity to the stream. A second set of factor scores is computed to reflect direct alterations to the flow regime from impoundments, flow diversions or discharges and levee banks. These factor scores are standardized and combined by linear weights. The values of the two resulting indices are then modified according to the level of disturbance to all upstream sections and their tributaries. Thus, undisturbed contributory streams can enhance the wild river potential of a stream while upstream disturbance reduces its potential.

These indices are combined in a composite River Disturbance Index (RDI) which gives an overall rating. However, all individual factor scores are retained for independent examination and analysis. All scores are standardized (range 0–1). A value at or near zero is at the undisturbed end of the continuum, while at or near one is at the severe disturbance end of the continuum.

Derivation of the RDI involves the application of a four-step procedure to every stream section (Fig. 1):

(i) Compute factor scores for key catchment condition and flow regime indicators by applying a grid-based GIS modeling procedure to the primary disturbance database.
(ii) For each stream section combine these factor scores to produce two indices: a Section Flow Regime Disturbance Index (SFRDI) which takes account of direct alterations to the flow regime such as dams, levee banks or flow diversions and a Sub-Catchment Disturbance Index (SCDI), which provides a measure of the indirect impact of human modifications in the sub-catchment.
(iii) Calculate the Catchment Disturbance Index (CDI) by accumulating upstream SCDI values adjusted by an estimate of the sub-catchments' relative contribution to total run-off; calculate a Flow Regime Disturbance Index (FRDI) by accumulating upstream SFDRI values adjusted by an estimate of the relative run-off accumulated at the points of disturbance.
(iv) Combine FRDI and CDI to form a RDI providing an overall index of potential river disturbance.

3. Application: characterizing the anthropogenic disturbance of Australian streams

The method requires a supporting database that records the nature and location of human activities, catchment and surface flow characteristics and a surrogate for run-off (Fig. 2). The development of this database and its use to characterize the anthropogenic disturbance pressure for Australian streams is described in the following sections.

An extensive review process was undertaken. Streams identified as least disturbed from a preliminary set of disturbance indices were evaluated at a series of workshops around Australia and by State water management and nature conservation authorities. A field based verification study was conducted in the Kimberley region of northwest Western Australia (Williams et al., 1996). Aerial inspection and limited on-ground truthing was used to assign rivers to one of five major categories, ranging from wild to degraded, based principally on evidence of erosion and sediment deposition. This study showed that selecting streams with low RDI scores gave a reasonably accurate indication of likely candidates for nomination as wild rivers. All aspects of both the method and its application were discussed and approved at regular meetings of a Project Advisory Committee with wide representation including relevant government agencies, landowners, indigenous peoples, the scientific community and conservation groups. In particular, the review process provided a forum for discussion on the weights to be adopted for particular disturbance categories and the types of data to be incorporated.

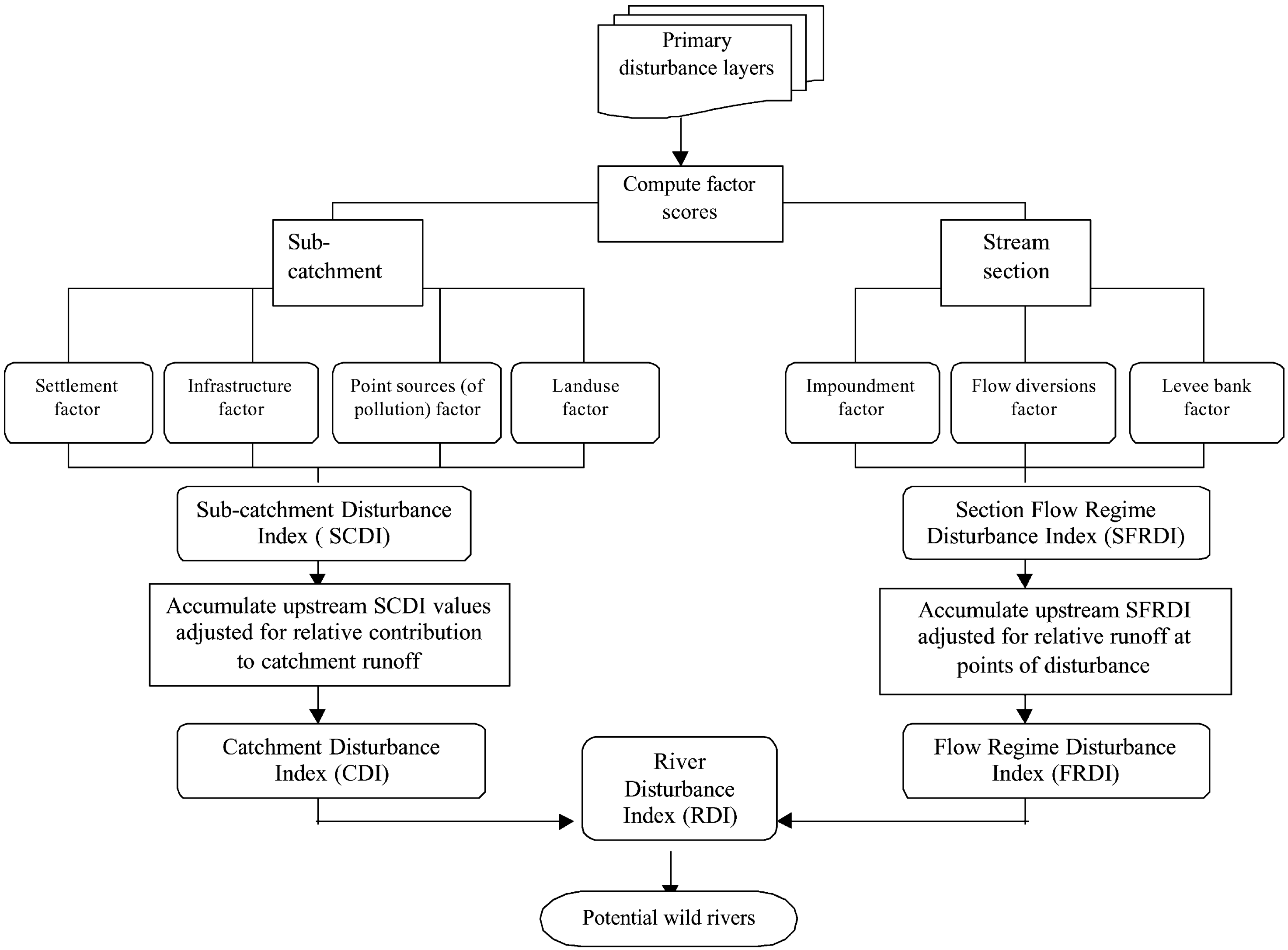

Fig. 1. The river disturbance model.

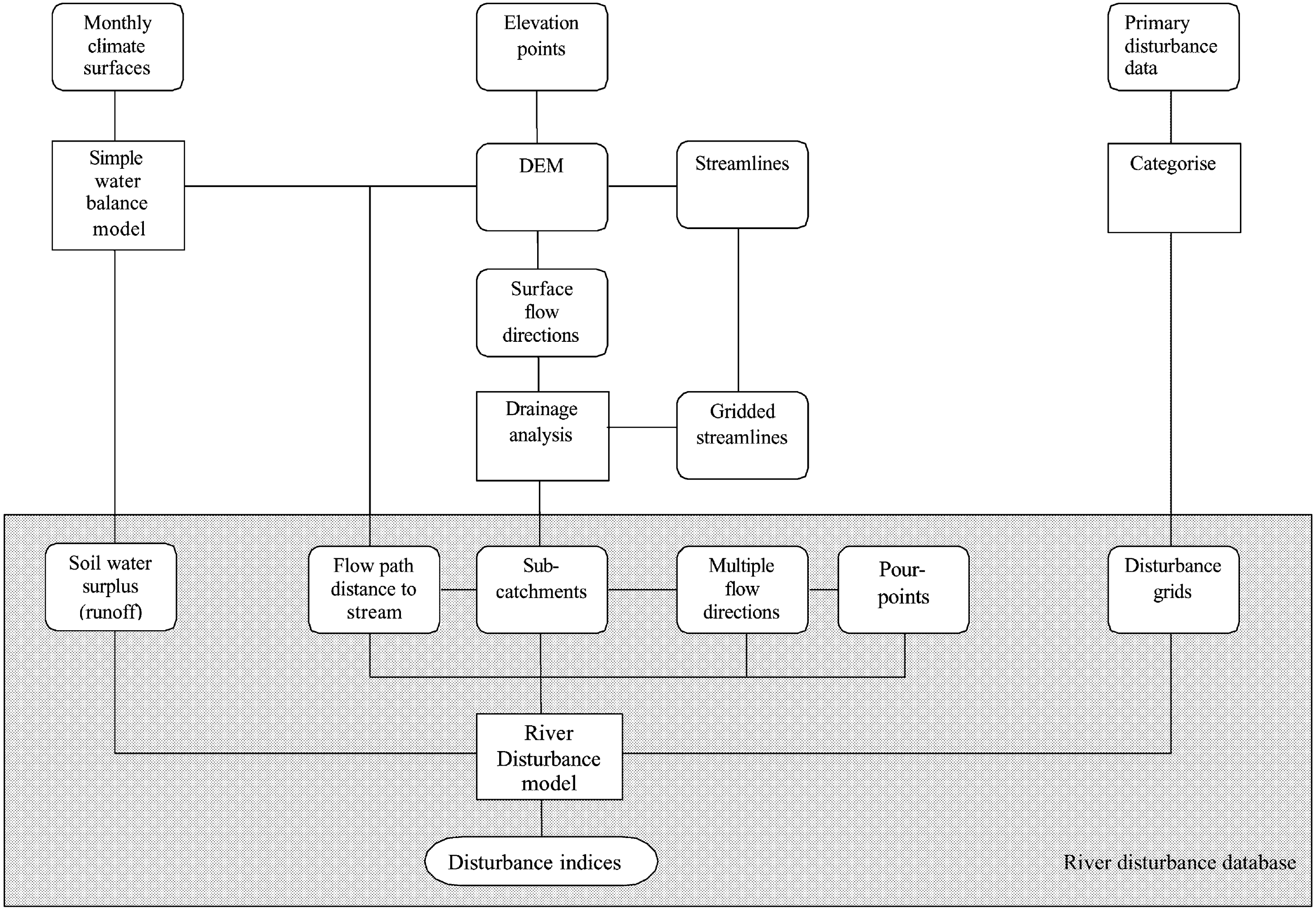

Fig. 2. Development of the river disturbance database.

3.1. Drainage analysis

An essential component of the procedure is the modeling of drainage paths and the delineation of sub-catchments for each stream section. Computer-based procedures based on analysis of a regular grid of elevation data (DEM) have been reviewed in Moore et al. (1991). The approach chosen here is based on the widely used, grid-based method of Jenson and Domingue (1988). Grids constitute an uncomplicated target for model programming and are readily compatible with other types of data (Holmgren, 1994).

3.1.1. The digital elevation model

A DEM with a spatial resolution of 250 m was computed from streamline, spot height and coastline data in the GEODATA TOPO-250K digital database (AUSLIG, 1992). The ANUDEM Program, Version 4.4 (Hutchinson, 1989, 1996) was used to derive the DEM from approximately 5 million irregularly spaced elevation points digitized from 1:100,000 scale topographic maps and natural stream courses from 1:250,000 topographic maps. Many errors in elevation, and streamline vectors not directed downslope had already been corrected in the production of a national 9 Second DEM (AUSLIG, 1997a). Drainage density and accuracy varied significantly between map sheets, reflecting both real topographic differences and cartographic interpretation. This variation is reflected in the representation of terrain features in the DEM and the derived drainage analysis. ANUDEM interpolates the elevation data by minimizing a suitably weak roughness penalty and by imposing constraints that ensure a connected drainage structure, directly from the streamline data (Hutchinson, 1996). By imposing a global drainage condition on the fitted grid, ANUDEM automatically removes the spurious sinks or pits that typify other DEM interpolation procedures. This significantly improves drainage analysis outputs.

3.1.2. Surface flow direction

Central to the task of both catchment delineation and modeling surface flow paths is the accurate assignment of drainage directions to individual grid cells. Costa-Cabral and Burges (1994) reviewed the principal methods for computing the drainage matrix from a DEM. The simplest and most widely used is the D8 method, where each grid cell discharges into one of its neighbors, the one located in the direction of steepest descent (Holmgren, 1994). This method is also compatible with widely used watershed delineation programs (Jenson and Domingue, 1988; ESRI, 1996).

The ANUDEM program produces a flow direction grid when fitting the DEM. This matrix follows the D8 convention but incorporates the information provided by the directed streamline input data, imposing associated side conditions. It was found to provide a better representation of flow paths in areas of low relief than would be achieved by simply allocating the direction of steepest descent.

Problems arise when grid cells with undefined drainage direction (i.e. a cell with no neighboring cells of lower elevation) remain in the flow direction grid. Because these cells do not contribute flow to any other cells, they act as sinks. Genuine sinks do occur, for example, in areas of aeolian deposition with dune and swale topography or ephemeral lakes and claypans (Marks, 1988). These geomorphic environments are widespread in the arid interior. In other areas, they are likely to be either artifacts of the DEM fitting process or, more commonly, due to data errors or inadequacies. One widely used corrective measure is that introduced by Jenson and Domingue (1988). The elevation of sink cells is raised to the height of the lowest outlet through which water will flow, i.e. the pour-point, and flow directions recomputed from the ‘filled’ DEM. However, using this automatic procedure, such as implemented in the ArcInfo FILL command, the directed streamline information incorporated into the flow direction matrix by ANUDEM is lost.

An alternative, but much more labor intensive, procedure was developed to systematically identify spurious sinks by reference to 1:100,000 or 1:250,000 scale topographic maps and then manually correct the flow direction of individual grid cells using the grid editing functions available in ArcInfo.

3.1.3. Catchment delineation

The WATERSHED function in ArcInfo GRID was used to determine sub-catchment areas for each stream section from the streamline coverages used in fitting the DEM. This function uses the flow direction grid to determine the contributing area above a set of grid cells or ‘seeds’ that represent the outflow points of the desired catchments. A commonly adopted method (Jenson, 1991) automatically initiates a seed at all

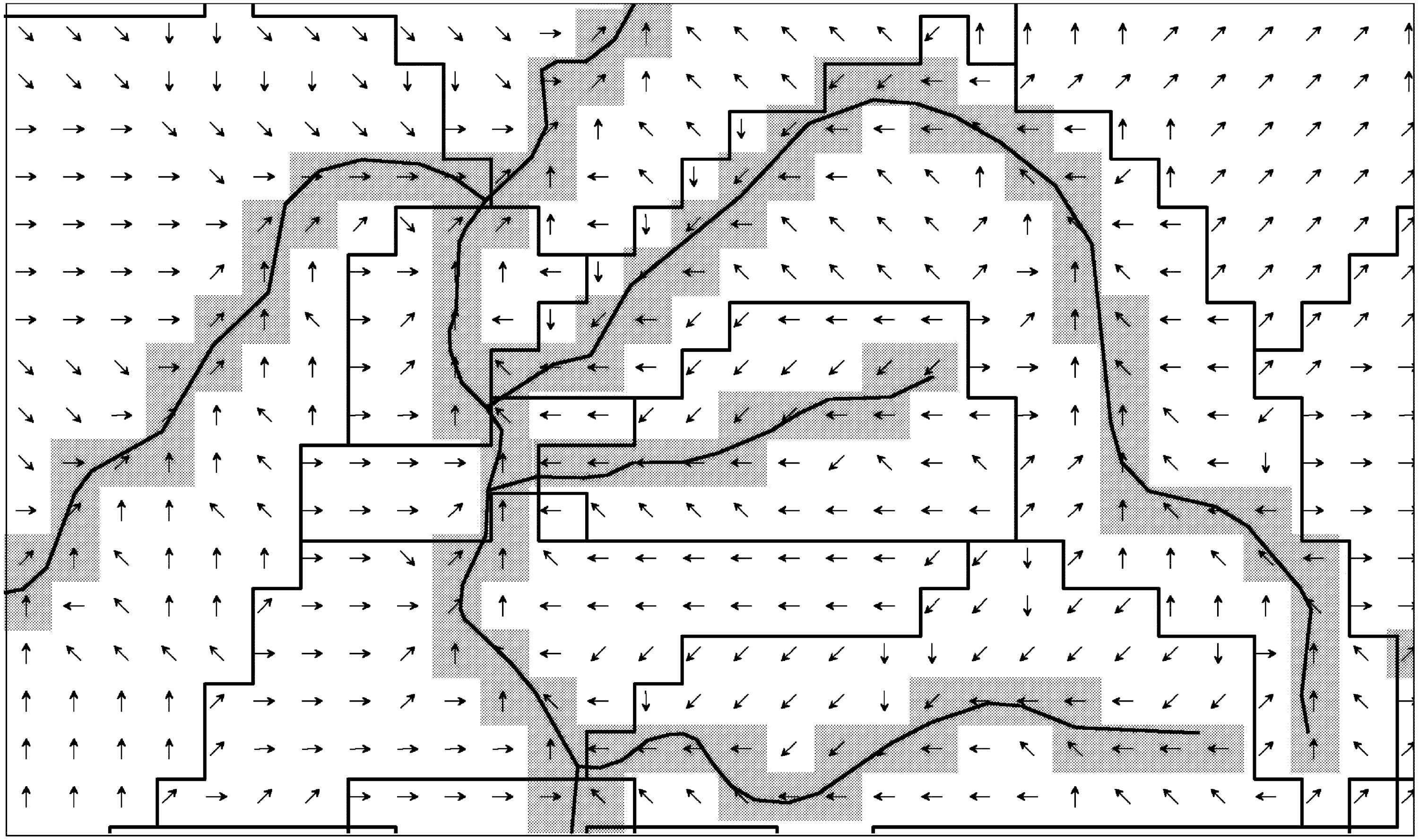

Fig. 3. Sub-catchment delineation. Sub-catchments are delineated by identifying all grid cells that contribute flow to the gridded representation of the streamline sections (shown in gray). Arrows indicate the flow direction. Bold lines are the sub-catchments delineated.

confluences of a drainage network which is determined by application of a user specified threshold to a flow accumulation grid (i.e. a count for each cell of the number of upstream cells that would contribute to it based on their flow directions). However, when tested, visual comparison with topographic mapping showed this method occasionally produced spurious channels, particularly in areas of low relief. Instead, the gridded representation of the mapped streamline network were used as seeds (Fig. 3).

Over 1,500,000 sub-catchments were defined for the mainland and 10,500 for the island state of Tasmania. The size of the delineated sub-catchments reflects mapped stream density. Accordingly, significant differences between neighboring map-sheets could reflect cartographic interpretation as much as real topographic differences. Nevertheless, the streamline database was the most accurate available with comprehensive national coverage and consistent topology, and provided an opportunity to consider smaller, but potentially important river sections.

3.1.4. Distributing flow between diverging channels

Stream networks that exhibit divergent flow characteristics, such as the extensively braided and anastomosing streams of the Channel Country of southwest Queensland, present particular difficulties in determining flow paths. Actual flows in the multiple channels may vary with every rainfall event and cannot be modeled with the data available. Nevertheless, the procedure used here dictated that an operational solution be developed. The D8 method of determining flow direction routes all flow from a grid cell to only one of its neighbors. Thus, where a stream branches into two or more channels, flow is routed down one channel only. The spatial resolution of the DEM (250 m) was too coarse to adopt methods such as that proposed by Holmgren (1994) which distribute flow proportionally to multiple neighboring grid cells relative to slope. An alternative procedure was developed to code the grid cells of a branching junction with multiple flow directions representing all directions to a neighboring grid cell of the downstream sub-catchment.

3.1.5. Sub-catchment pour-points

The outlet or pour-point of a sub-catchment is the grid cell(s) with a flow direction out of the sub-catchment. However, inconsistencies between the gridded streamlines and the flow direction grid can result in some pour-point cells receiving accumulated flow from other than upstream tributaries (Fig. 4). As the CDI and FRDI values are computed from the accumulated values at pour-points this could reduce the wild river value of an undisturbed tributary where it incorrectly received flow from another more disturbed tributary. To overcome this a procedure was developed to locate pour-point cells that: (1) received upstream flow (unless a first order stream); and (2) did not receive flow from streams sharing the same downstream section.

If a grid cell that met these criteria could not be located, the grid cell with the highest accumulated flow in the sub-catchment was selected as a pour-point. In the particular case of branching or diverging channels, all grid cells with a flow direction out of the sub-catchment were selected.

3.2. Compiling the catchment and river disturbance layers

Source data were generally compiled at a scale of 1:250,000 over a period from 1989 to 1997 although in some areas data were compiled at scales of 1:500,000, 1:100,000 or 1:50,000. These differences in scale affect the ability to reliably determine intersections between different data layers, for example, as roads and streams were digitized from maps of different scale, road crossings could not be confidently identified.

Major sources of data were as follows:

1. The National Wilderness Inventory (NWI) primary database (Lesslie and Maslen, 1995). This included the location of a range of settlement and infrastructure features, the extent of non-natural land cover and an index of biophysical naturalness (BN). Additional data from state government agencies were used to selectively update these coverages for this analysis.
2. The GEODATA TOPO-250K database (AUSLIG, 1992) produced from the 1:250,000 scale topographic map series. Included were the location of built-up areas, infrastructure, reservoirs and canals.

The large number of categories for each feature type within these databases were reduced to a smaller set by combining those categories that were likely to have a similar magnitude of impact on a stream. Full

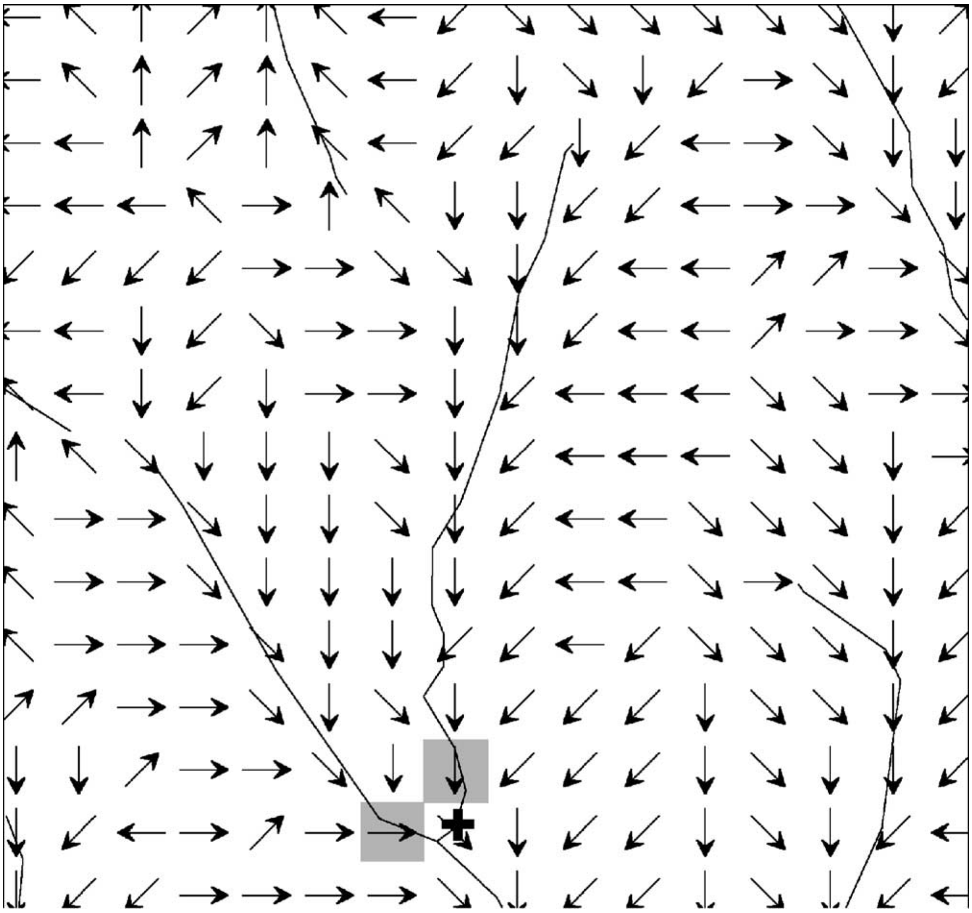

Fig. 4. Identification of sub-catchment pour-points. The cell indicated with a '+' was not used as a pour-point as it receives flow from a tributary of the same order.

details of the data sets used and the translation of codes to the new data categories are given in Stein et al. (1998).

Each category was assigned a weight, such that each feature was represented in equivalent units to the feature with the greatest impact. This approach follows a similar method used for the NWI (Lesslie and Maslen, 1995) but with weights assigned according to their potential impact on aquatic ecosystems rather than on wilderness values. A distance threshold was also set for each point source category to allow the values ascribed to these impacts to be modified according to their proximity to the stream. Beyond the threshold, the values are reduced according to a simple distance decay function. Initial values for weights and distance thresholds were determined from literature review and professional judgment to reflect potential impact on stream ecological, geomorphological and hydrological processes, relative to the category with the greatest impact. These weights and thresholds were then reviewed both by the Project Advisory Committee, government land and water management and nature conservation authorities.

Primary disturbance data for each data type, whether polygon, line or point, were subdivided according to the data categories defined for each factor, and converted to grids consistent with the 250 m DEM. All grids were coded 1/0 for presence/absence of a feature in each data category. When converting from vector data, grid cells were assigned the value of that polygon feature which covered the largest area of the grid cell or, in the case of line features, the line feature with the greatest length in the grid cell. Thus, to ensure that no categories were missed, separate grids were produced for every data category. Where more than one feature of a data category occurred in a grid cell, the cell was still assigned a value of 1.

3.3. Computing the river disturbance indices: sub-catchment and stream section factors and indices

Factor scores are computed for each sub-catchment or stream section by overlaying the sub-catchment grids with the disturbance layers. Classical overlay problems could occur where data were derived from more than one source and/or from maps at different scales, for example, the same feature could be represented by more than one grid cell simply as an artifact of slight differences in its location between data sources. It was not always possible to determine which of the data sources were most reliable, so a factor score was computed from each available data source and the maximum score assigned to a sub-catchment or stream section. It was more likely that a given dataset missed features than included non-existent features.

3.3.1. The distance decay function

The value ascribed to discrete impacts, i.e. settlements, infrastructure, levee banks and point sources of pollution, was modified according to proximity of the impact to the stream using a simple distance decay function. The Distance Modifier (DM) is calculated as

$$\mathrm{DM} = 1, \quad \text{for impacts closer than threshold to stream}$$

otherwise

$$\mathrm{DM} = \frac{\text{threshold}}{\text{surface distance from stream}}$$

The distance function is applied by multiplying the weighted value for each feature by the calculated DM for all grid cells in the sub-catchment.

The distance from stream was calculated for each grid cell as the flow path length across the surface using the ArcInfo GRID function FLOWLENGTH. The arguments for this function were: (1) a weight grid, computed as the difference in elevation per meter moved horizontally, from a cell to its downstream neighbor; and (2) the flow direction grid, modified such that all streamline grid cells were set to an undefined flow direction value and, thus, interpreted as a sink. Streamline grid cells were then assigned a distance to stream value of 1 to avoid an undefined value for the distance function.

3.3.2. Settlement factor

Settlements act in many ways to alter the hydrological, geomorphological and ecological characteristics of streams and so are considered as a separate component of the SCDI. For example, increased soil erosion may result in increased sedimentation, especially during construction phases (Harbor, 1999). Large areas of impervious surface leads to increased incidence of run-off events and greater magnitude of peak discharges (Crabb, 1997). This run-off may contain significant pollutants (Gardner et al., 1997; Calace et al., 2001) including organic pollution from exotic riparian species (Pen, 1999) with significant implications for stream ecological processes.

Five categories of settlement or structure were defined in the original source data (Table 1). Each category could be assigned a weight commensurate with its potential level of impact on a stream. The weightings used in this study reflect the increasing likelihood of greater impact with increased intensity of urbanization. For example, a large city is likely to produce the greatest impacts on stream hydrology with its higher proportion of paved areas, a greater impact on water

Table 1
Weights and distance thresholds for settlement data categories

Category number	Description	Weight	Distance threshold (m)
1	Major grade (e.g. service location >100 persons)	1.0	5000
2	Intermediate grade (e.g. service location 10–100 persons)	0.8	2500
3	Minor grade (e.g. service location <109 persons)	0.6	1500
4	Residential (e.g. farmhouse, holiday cottage)	0.1	750
5	Other structures (e.g. tower, windmill, trig station)	0.1	500

quality from contaminated storm-water run-off and is more likely to have altered the geomorphic condition of the stream through channelization and removal of riparian vegetation. All weights are, thus, scaled relative to a major grade settlement with a maximum value at 1.

At each grid cell in which a settlement or structure occurs, a settlement score was calculated as the weight assigned to the settlement category multiplied by the value of the distance decay function computed for the appropriate distance threshold (Table 1). Where more than one settlement data category occurred within a grid cell, the category with the highest weight was used. A Settlement Factor (SF) score was calculated for each sub-catchment by summing the grid cell scores. This score was standardized by dividing by the potential maximum for the sub-catchment, that is, the sum of the major grade settlement weights multiplied by the DMs for a major grade settlement (Fig. 5). The score, thus reflected both the density of settlement within the sub-catchment and its proximity to the stream.

3.3.3. Infrastructure factor

Infrastructure has a variety of impacts on the stream including increased sedimentation, altered run-off characteristics and changes in water quality. Road construction may result in significant loss of biodiversity due to restricted movement between populations, increased mortality, habitat fragmentation and edge effects, invasion by exotic species, or increased human

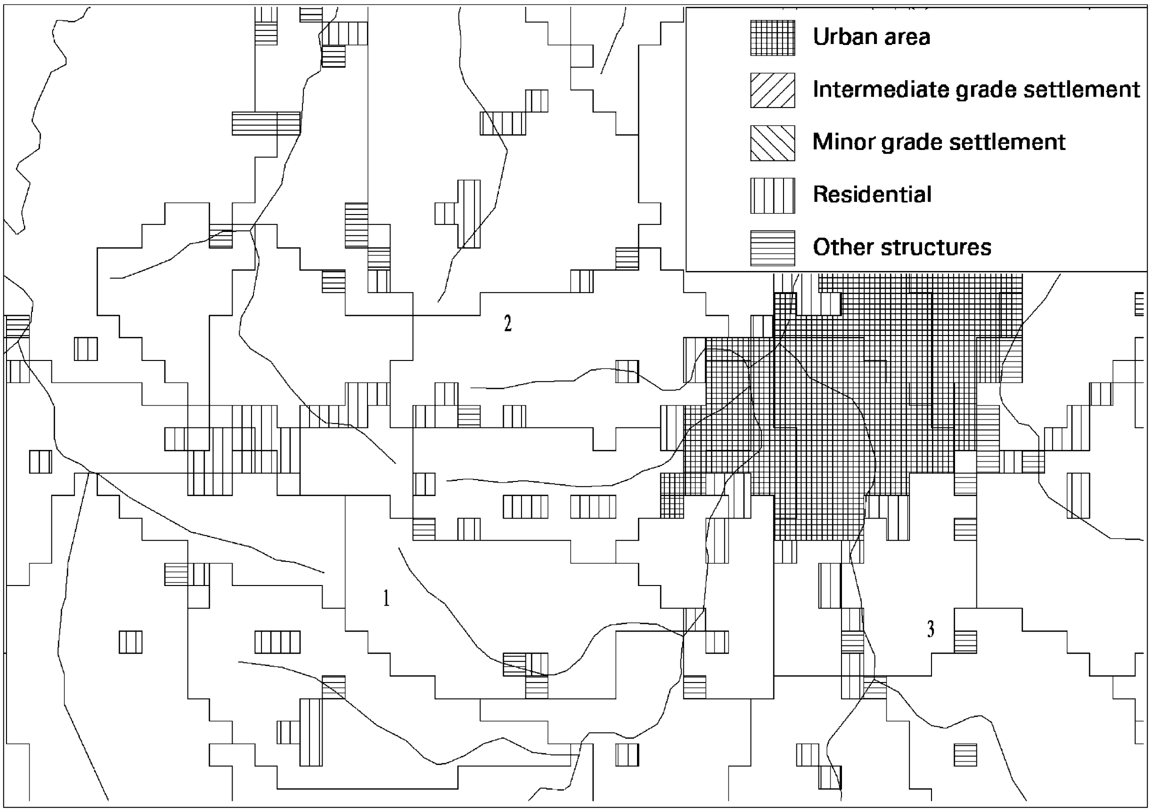

Fig. 5. SF example: grid cells with a value of one in the settlement data grids are shown. Sub-catchment boundaries are drawn for each stream section. The sub-catchment numbered 1 has a very low density of settlement with only three grid cells with either settlement category 4 or 5 (residential property or other structures) and has a computed SF value of 0.005. The sub-catchment numbered 2 also has residential grade and other settlements but includes the edge of the urban area (settlement category 1). It has a SF value of 0.029. About half of the sub-catchment numbered 3 is an urban area which is assigned the highest weight and most of the member grid cells are within the 5 km distance threshold (Table 1) so the SF value is much higher (0.445).

Table 2
Weights and distance thresholds for infrastructure categories

Category number	Description	Weight	Distance threshold (m)
1	Main sealed road	1.0	1000
2	Other sealed road	0.8	750
3	Unsealed road	0.8	500
4	Railways	0.8	500
5	Vehicular track	0.4	250
6	Utilities	0.2	500
7	Walking track	0.01	250

access to aquatic habitats (Findlay and Bourdages, 2000). Seven categories of infrastructure were defined within the national data coverage. Each category could be weighted according to its estimated impact on a stream. In this case, all weights were scaled relative to a main sealed road set at a value of 1 (Table 2). While unsealed roads and tracks may contribute significantly to stream turbidity (Croke et al., 2001) the usually higher volume of traffic on sealed roads leads to a higher likelihood of contaminated run-off from spillages, petrol additives, etc. (Trombulak and Frissell, 2000). More significant alterations to the timing and routing of run-off and groundwater flow are also expected as a result of the major earthworks involved in the construction of main roads. Even walking tracks in natural areas have some potential for stream disturbance through increased erosion and establishment of exotic weeds (Jesson et al., 2000) but these impacts are likely to be orders of magnitude less than other infrastructure. The distance function was applied at each grid cell using the distance thresholds (Table 2), before summing across the sub-catchment and standardizing. Forman and Alexander (1998) defined the 'road effect zone', the maximum distance at which significant ecological effects of roads occur. This zone was estimated to vary between 200 and 800 m in the United States (Forman, 2000), values consistent with the distance thresholds used here.

Thus, final Infrastructure Factor (IF) score reflected the greater impact of, for example, a major freeway close to a stream compared to a track some kilometres distant from a stream. For any grid cells coded with more than one infrastructure category, such as where a track crosses a main sealed road, the value assigned was the maximum of the weight multiplied by the DM for each of the possible categories.

3.3.4. Point sources (of pollution) factor

Many human activities produce point sources of pollution. Industrial activity has long been identified as a major source of contaminants for aquatic environments via atmospheric deposition and wastewater discharge (Ali and Sreekishnan, 2001). Release of contaminated water from mining or mineral processing can lead to extensive contamination for considerable distances downstream of the site (Kelly, 1988; Farrell and Kratzing, 1996; Watanabe et al., 2000). Point sources producing nutrient rich discharges include abattoirs, dairies, feedlots, fish farms, piggeries, poultry farms, saleyards and sewerage treatment works. Nutrient enrichment may have significant effects on both the abundance and community composition of aquatic biota, for example, Donnelly et al. (1999) reported algal blooms and the collapse of macrophyte populations associated with the input of treated sewage effluent. The actual impact on aquatic biota and stream ecosystem processes will be related to the contaminant load and the varying tolerances of the biota, but such information was unavailable for other than a small number of sites or species. Thus, assignment of weights and distance thresholds was based on professional judgment with extensive review by advisory committee members and state agency representatives.

To calculate a Point Sources Factor (PSF) score each category of point source was assigned a weights scaled relative to the potential impact of an operating mine set at a value of 1, modified by a distance factor to account for proximity to stream (Table 3). In contrast to other sources of catchment disturbance, the potential impact of point sources is not related to their areal

Table 3
Weights and distance thresholds for point source (of pollution) categories

Category number	Description	Weight	Distance threshold (m)
1	Operating mine	1.0	5000
2	Contaminated site	1.0	5000
3	Point pollution (chemical)	1.0	5000
4	Point pollution (nutrient)	1.0	1000
5	Petroleum well	0.4	1500
6	Quarry	0.4	750
7	Abandoned mine/quarry	0.2	750
8	Petroleum well (abandoned)	0.1	500

Table 4
Weights for land use categories

Category number	Land use category[a]	Weight
Cultural land cover		
1	Urban, cropping, irrigated cropping	1.0
2	Plantations, livestock grazing	0.75
Natural land cover		
3	Clear-fell logging operations and/or intensive grazing	0.5
4	Repeated selective logging and/or light-moderate grazing	0.35
5	Selective single logging and/or irregular grazing	0.2
6	Logged or grazed historically (not within preceding 60 years)	0.05
7	Unlogged and ungrazed	0.0

[a] Classes 3–7 equate with the five classes of the BN index, a rating procedure related to the intensity and duration of grazing by livestock and the harvesting of timber, derived for areas with natural land cover (Lesslie and Maslen, 1995). The class descriptions given here are for the 'non-arid' areas where grazing is essentially unrestricted by the availability of water and commercial timber harvesting may take place. Where arid or semi-arid livestock grazing predominates the rating system depends on the range-type, land tenure and access to (semi) permanent water and produces classes comparable in grazing intensity.

extent in the sub-catchment, so PSF scores were standardized simply by setting to a maximum value of 1.

3.3.5. *Land use factor*

The Land Use Factor (LUF) incorporates seven classes of diffuse (area-based) impacts (Table 4). The impacts of catchment land uses include changes to the sediment and flow regimes, increasing nutrient supply and salinization, and heavy metal, toxic chemical and thermal pollution (Boulton and Brock, 1999). A LUF was calculated as the proportion of the sub-catchment with land cover altered by post-European settlement, weighted (Table 4) according to the relative impact of the land use on the stream. Cropping and urban land uses were assigned the highest weight because of their greater likelihood of contaminated run-off (e.g. Gilliom, 2001) and major alterations to geomorphological and hydrological regimes associated with the replacement of natural land cover (Crabb, 1997). Grazing by introduced ungulate stock is a major land use over much of Australia with increased grazing intensity associated with a decline in water quality and riparian condition (Harding et al., 1999; Jansen and Robertson, 2001). The other major broad area land use considered in the LUF is forestry. The impacts on streams include changes to water quality, for example, elevated turbidity and sedimentation accompanying timber harvesting activities and track construction and increased nutrient concentrations following fertilizer applications (Nisbet, 2001). Increased stream flow is commonly reported immediately following logging followed by yields below pre-logging levels in rapidly regenerating forests (e.g. Lane and Mackay, 2001). An intact riparian zone may buffer the stream against the adverse impacts of catchment land use (Franklin, 1992; Malanson, 1993). Unfortunately, no data were available to discriminate according to the proximity of the land cover disturbance to the stream, or the condition of the riparian zone.

3.3.6. *Impoundment factor*

Dams, weirs and other in-stream structural barriers break the natural continuum of a river, converting it into a sequence of isolated sections (Young et al., 2001). They impact directly on aquatic biota, preventing movement and fragmenting populations (Jager et al., 2001). Indirect impacts arise from alterations to the nature and transport of particulate organic and inorganic matter (Kelly, 2001) and the hydrological and geomorphological behaviour of the stream (Baxter, 1977; Petts, 1984; Thoms et al., 1999; Sheldon et al., 2000; Peters and Prowse, 2001; Maingi and Marsh, 2002). The Impoundment Factor (IMF) was computed simply as the maximum of the weights (Table 5) of any impoundment features recorded for a stream section. There are little data available to compare the impacts of various types of in-stream structures. Even minor structures may be barriers to some species (Jackson, 1997). However, the effects of weirs are more localized than dams (Walker et al., 1992) so weights were scaled relative to large dams. Stream sections with no

Table 5
Weights for impoundment categories

Category number	Description	Weight
1	Major structure	1.0
2	Weir	0.3
3	Locks/sluice gates	0.3
4	Minor structure	0.1

impoundments of any type were assigned an IMF value of zero. Off-stream impoundments, including farm dams, were taken into account only when they fell within a grid cell also coded as a streamline. However, it is recognized that the proliferation of farm dams, in addition to the recent development of pumped off-stream structures based on flood flows, are likely to have a significant impact on in-stream flows, especially in inland river systems, and will need to be considered in future revisions of this assessment.

3.3.7. *Flow diversions/alterations factor*

A Flow Diversions/alterations Factor (FDF) reflects the impact of alterations to the flow regime due to diversion or enhancement of stream discharge. The presence of constructed channels for water supply, hydroelectric power generation, irrigation and drainage provides a surrogate indicator for these flow alterations. Donor rivers experience changes associated with reduced discharge (Kingsford, 2000; Young et al., 2001) while recipient systems undergo changes in the timing, magnitude and velocity of flow with resultant geomorphological and ecological changes (Gibbins et al., 2000). In particular, interbasin transfer schemes may lead to changes in major ion and nutrient concentrations, altered thermal characteristics and pH, and immigration of exotic species with serious implications for biotic integrity (Stanford and Ward, 1992; Davies et al., 2000). Drains are also responsible for increasing delivery of pollutants to the stream (Pen, 1999).

The limited number of data categories used (Table 6) reflect the available data sources. Only data recording the geographic location of a channel or drain were generally available. As a result, it was not possible to categorize channels according to their function or the magnitude or direction of the changes to stream discharge. FDF was assigned as the maximum of the weights assigned to the classes of flow diversion recorded for a stream section (Table 6). For example, a stream section intercepted by a canal (category 2) would be assigned a value of 0.6. This factor has a value of zero for stream sections where no flow diversions or extractions were identified.

Table 6
Weights for flow diversion data categories

Category number	Description	Weight
1	Major (e.g. penstock, water pumping station)	1.0
2	Other (e.g. channel, canal)	0.6

3.3.8. *Levee bank factor*

Levee banks are constructed to reduce the risks of flooding on human populations or to redirect water to off-river storages. Levees effectively increase channel depth and, thus, stream power with implications for channel geomorphic behavior (Pen, 1999). They reduce the size of the effective floodplain, reducing the amount of floodplain resources available to the river in high flow events (Young and Hillman, 2001). Reduced flooding may lead to changes in the composition, growth and regeneration of floodplain vegetation and reduce wetland breeding and refuge habitat (Kingsford, 2000). To compute the Levee Bank Factor (LBF) a distance function was applied using a distance threshold of 500 m, to reduce the influence of levee banks distant from a stream. The score assigned to the stream section was the maximum, distance modified, value within the sub-catchment.

3.3.9. *Sub-catchment and section indices*

For each stream section a SCDI was computed as the weighted sum of the four catchment-based factors; settlement, infrastructure, point sources of pollution and land use:

$$\mathrm{SCDI} = \frac{(W_1 \times \mathrm{SF}) + (W_2 \times \mathrm{IF}) + (W_3 \times \mathrm{PSF}) + (W_4 \times \mathrm{LUF})}{W_1 + W_2 + W_3 + W_4}$$

where SF is the settlement factor; IF the infrastructure factor; PSF the point sources factor; LUF the land use factor; and $W_1, \dots, W_4$ are the optional weights set as follows: $W_1 = 0.25$, $W_2 = 0.12$, $W_3 = 0.13$; $W_4 = 0.5$.

Each factor contributes independently to SCDI although it is recognized that these are not truly independent in a statistical sense. However, the nature of interactions between them is highly complex (Townsend and Riley, 1999) and location and region specific. Again, these weights were assigned initially by professional judgment and literature review. There is little quantitative evidence in the literature for the relative importance of each of these factors but leaving them unweighted effectively assigns each a weight of 0.25. Extensive consultation confirmed a preference for the higher weighting for land use which has a broader range of impacts on streams. In contrast, point sources of pollution may have serious consequences for aquatic biota but no impact on stream hydrological or geomorphological characteristics. This index only records disturbance within the immediate sub-catchment of a stream section and so may have low values indicating little disturbance even where contributory streams are highly disturbed.

Similarly, a SFRDI was computed as the weighted sum of the factors associated with direct alterations to the flow regime. This index is, thus, non-zero only in those sections of a stream with a structure that alters the flow in some way, for example, a dam or weir:

$$\mathrm{SFRDI} = \frac{(W_1 \times \mathrm{IMF})+(W_2 \times \mathrm{LBF})+(W_3 \times \mathrm{FDF})}{W_1 + W_2 + W_3}$$

where IMF is the impoundment factor; LBF the levee bank factor; FDF the flow diversions/alterations factor and; $W_1, \ldots, W_3$ are the optional weights set as follows: $W_1 = 0.4$, $W_2 = 0.2$, $W_3 = 0.4$.

3.4. Computing the river disturbance indices: incorporation of upstream disturbances

CDI values were computed for each sub-catchment to reflect the level of disturbance within the total catchment. CDI was based on the SCDI values of all contributory sub-catchments adjusted so that the contribution of any sub-catchment was proportional to its potential contribution to catchment run-off. Similarly, FRDI values were computed for each stream section by accumulating SFRDI values for all contributory stream sections.

No direct measures of run-off were available for all Australian streams and their tributaries and certainly not at the level of spatial resolution in this study. A relative scaling measure across the whole continent was derived from a simple water balance model coupled to long-term weekly mean rainfall and potential evaporation data. The water balance is a component of the plant growth model, GROWEST (Nix, 1981) as implemented in the GROCLIM sub-program of ANUCLIM (McMahon, 1995). In this simple tank model, rainfall is added to previous soil water storage and evapotranspiration removes water. Rainfall exceeding 'tank full' may be lost through run-off and/or deep drainage. In this case, the annual mean water surplus is used as a surrogate for run-off. Typically, a default value of 100 or 150 mm maximum plant available soil water storage is used to characterize better agricultural and pastoral soils. However, when using long-term weekly mean precipitation and potential evaporation values, a 50 mm maximum soil water storage was found to generate an accumulated surplus, and hence, run-off, that was congruent with long-term run-off measures at sample stations. Actual run-off is a complex function of many other factors. These include topography, geology, soil characteristics, rainfall intensity and duration, land cover and land use. Very high rainfall variability in Australia generates patterns of run-off that reflect extreme events rather than long-term average rainfall. Nevertheless, at regional or national scales, the simple water balance and catchment area are the dominant source of spatial variation in observed flow. This relationship was tested with data obtained from the Victorian Water Resource Data Warehouse (see http://www.vicwaterdata.net). Only gauging stations on unregulated streams (those with FRDI values equal to zero) were considered, thereby excluding the large streams with higher flow. A square root transformation was applied to reduce the data skew. For these 117 guaging stations, average daily flow was found to be strongly correlated with the accumulated soil water surplus ($R^2 = 0.93$, $P < 0.001$) (Fig. 6).

At each grid cell, the SCDI value was multiplied by the soil water surplus estimate. Where the mean annual soil water surplus estimate was zero, a value of 1 was assigned, so that the relative contribution of such sub-catchments to CDI was proportional to sub-catchment area only. These 'adjusted' SCDI values were then accumulated for all upstream grid cells using a program analogous to the FLOWACCUMULATION function in ArcInfo, but allowing for multiple flow directions. The CDI was then computed for each sub-catchment as the sum of the accumulated values at the pour-point grid cells, standardized by division by the accumulated soil water surplus. Thus,

$$\mathrm{CDI} = \frac{\sum_{i=1}^{n} \mathrm{ACCUMSCDI}_i + \mathrm{SCDISW}_i}{\sum_{i=1}^{n} \mathrm{ACCUMSW}_i + \mathrm{SW}_i}$$

where CDI is the catchment disturbance index; SW the soil water surplus; SCDISW the SCDI × soil water surplus; ACCUMSCDI the sum of SCDISW values

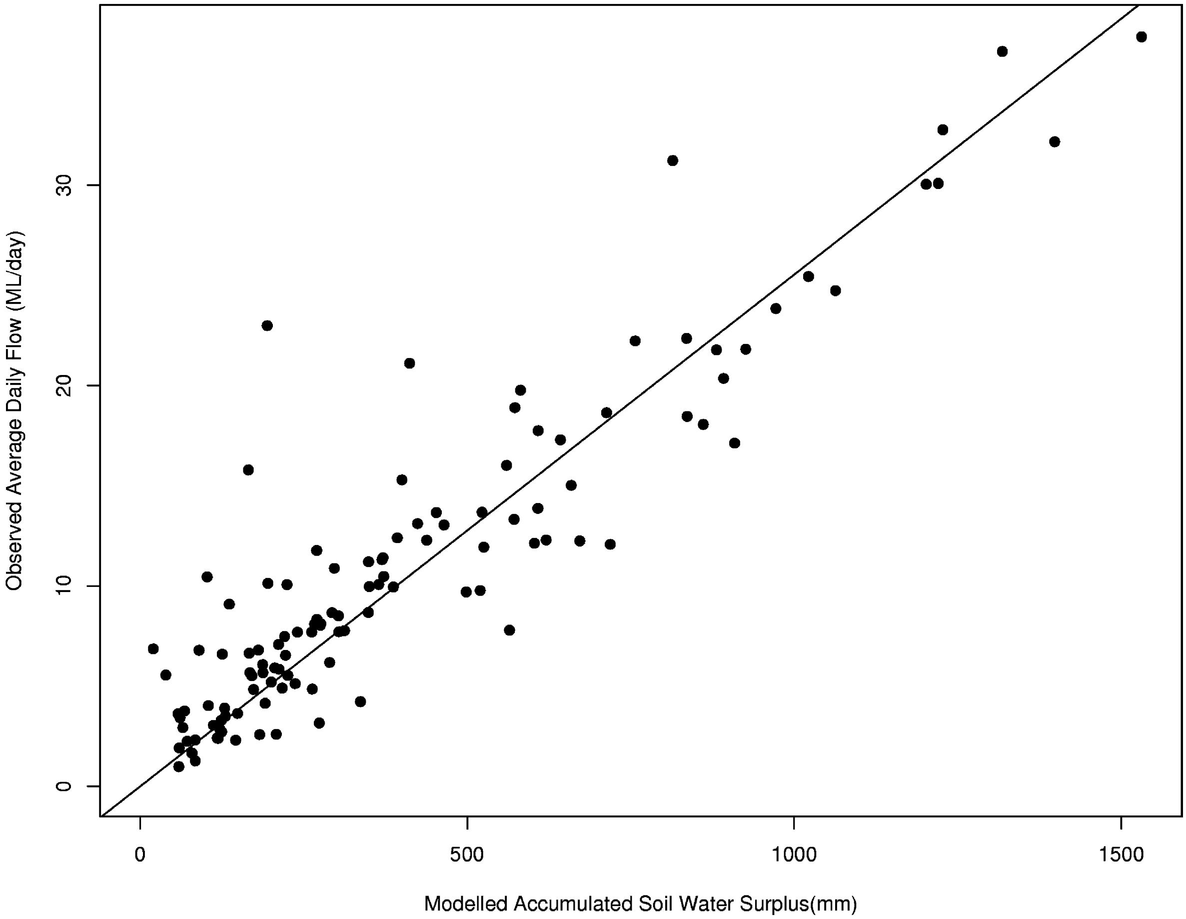

Fig. 6. Relationship between the accumulated soil water surplus, used as a surrogate for run-off, and average daily flow values for gauging stations on unregulated streams (FRDI equals zero) from the state of Victoria. Nation-wide, the accumulated soil water surplus values range up to 64,156,544 mm. Plot shows square root transformed values and fitted regression line.

for all upstream grid cells; ACCUMSW the sum of soil water surplus values for all upstream grid cells; and n is the number of pour-point grid cells in the sub-catchment.

SFRDI values were also accumulated for all upstream sections. However, values were adjusted for the total accumulated upstream soil water surplus, not just for that in the immediate sub-catchment. This distinction was made as lower SFRDI values upstream should not 'dilute' the impact of flow modifications such as a dam. Again, final FRDI values were standardized by dividing by the sum of the soil water surplus values for all upstream grid cells. For any grid cell, the accumulated product of SFRDI and the accumulated soil water surplus were constrained by the value of the accumulated soil water surplus, so that the ratio, FRDI, would not exceed 1. Thus,

$$\mathrm{FRDI} = \frac{\sum_{i=1}^{n}\mathrm{ACCUMSFRDI}_i + \mathrm{SFRDISW}_i}{\sum_{i=1}^{n}\mathrm{ACCUMSW}_i + \mathrm{SW}_i}$$

where SW is the soil water surplus; SFRDISW the SFRDI $\times$ ACCUMSW; ACCUMSFRDI the sum of SFRDISW values for all upstream pour-point grid cells constrained to never exceed ACCUMSW for any grid cell; ACCUMSW the sum of soil water surplus values for all upstream grid cells; n is the number of pour-point grid cells in the sub-catchment.

Stream networks that exhibited divergent flow characteristics were treated as a special case, where accumulated upstream values were apportioned between diverging channels, according to the multiple flow directions assigned to these grid cells.

3.5. The river disturbance index (RDI)

The RDI combines the CDI and FRDI to give an overall indication of the extent to which natural river processes might have been degraded. It was computed as the simple weighted average of CDI and FRDI:

$$\mathrm{RDI} = \frac{(W_1 \times \mathrm{CDI}) + (W_2 \times \mathrm{FRDI})}{W_1 + W_2}$$

where FRDI is the flow regime disturbance index; CDI the catchment disturbance index; and W_1 and W_2 are the optional weights set as follows: $W_1 = 0.5$, $W_2 = 0.5$.

Clearly, this formulation assumes that in-stream and catchment disturbances impact equally and independently on the river. Of course, differential weightings could be used and indeed, may be warranted in different environments. However, given the need for a standardized national overview equal weightings were used.

4. Results

More than 1.5 million stream sections with a total length of over 3×10^6 km were assessed. A list of essentially undisturbed or potential wild rivers was identified from those streams with a RDI score of not more than 0.01 (Stein et al., 2001). Just 19% or 591,332 km of the stream length satisfied this criterion. More than three quarters of this undisturbed length was in small streams or headwater tributaries with a catchment area of less than 10 km^2; very few large rivers remain undisturbed. Over 80% of the undisturbed stream length fell within the monsoonal tropical north or the arid/semi-arid center of the continent. The final wild river status of these rivers has now been verified by state agencies with responsibility for river management (see http://www.heritage.gov.au/anlr/code/arcmaps.html for details).

Detailed interpretation of disturbance index values for more disturbed streams will rely on evaluation of the relationship between these values and measures of river health/condition. Nevertheless, a useful national summary of the extent and the potential impact of human activities is possible using the reporting framework of the 244 river basins of Australia (AUSLIG, 1997b).

A disturbance profile was produced for each river basin using the decile values computed from the RDI/cumulative stream length curves. Basins were then grouped according to the similarity of their disturbance profile using a hierarchical agglomerative clustering technique in the PATN software package (Belbin, 1993). Dissimilarity was computed using the Bray–Curtis measure (Bray and Curtis, 1957) and grouping undertaken with the UPGMA fusion strategy (Sokal and Michener, 1958).

Nine groups were identified from the dendrogram. One of these groups comprised one small basin only, the Whitsunday Islands with a total stream length of just 128 km, for which there was almost no recorded disturbance. This single basin group was combined with group 1 which comprised the other five river basins largely undisturbed by European settlement, all of which are located in northern Australia. Groups 2–6 represent river basins with low to moderate levels of disturbance, while groups 7 and 8 include those basins with high levels of disturbance and in which almost all streams are disturbed to some extent (Table 7). Almost half of the 244 basins, with a quarter of the total stream length, fell within the two most disturbed groups. Not surprisingly, these basins occupy the more densely populated and/or intensive agriculture zone in Australia (Fig. 7).

The scarcity of large undisturbed river systems is highlighted when RDI values are summarized according to the upstream catchment area of a stream section (Table 8). While 21% of small headwater streams with catchment areas less than 100 km^2 remain undisturbed, almost all large rivers (catchment area greater than 5000 km^2) are potentially disturbed due to the cumulative impacts of human activities on downstream reaches. If only direct, in-stream alterations to the flow regime are considered, the results for large rivers are equally bleak. Nearly two-thirds of the stream length with catchment area greater than 5000 km^2 have FRDI values greater than 0.01, indicating they are affected to some extent by dams, flow diversions or levee banks.

The results have been presented here as national summaries. However, all factor and index scores are

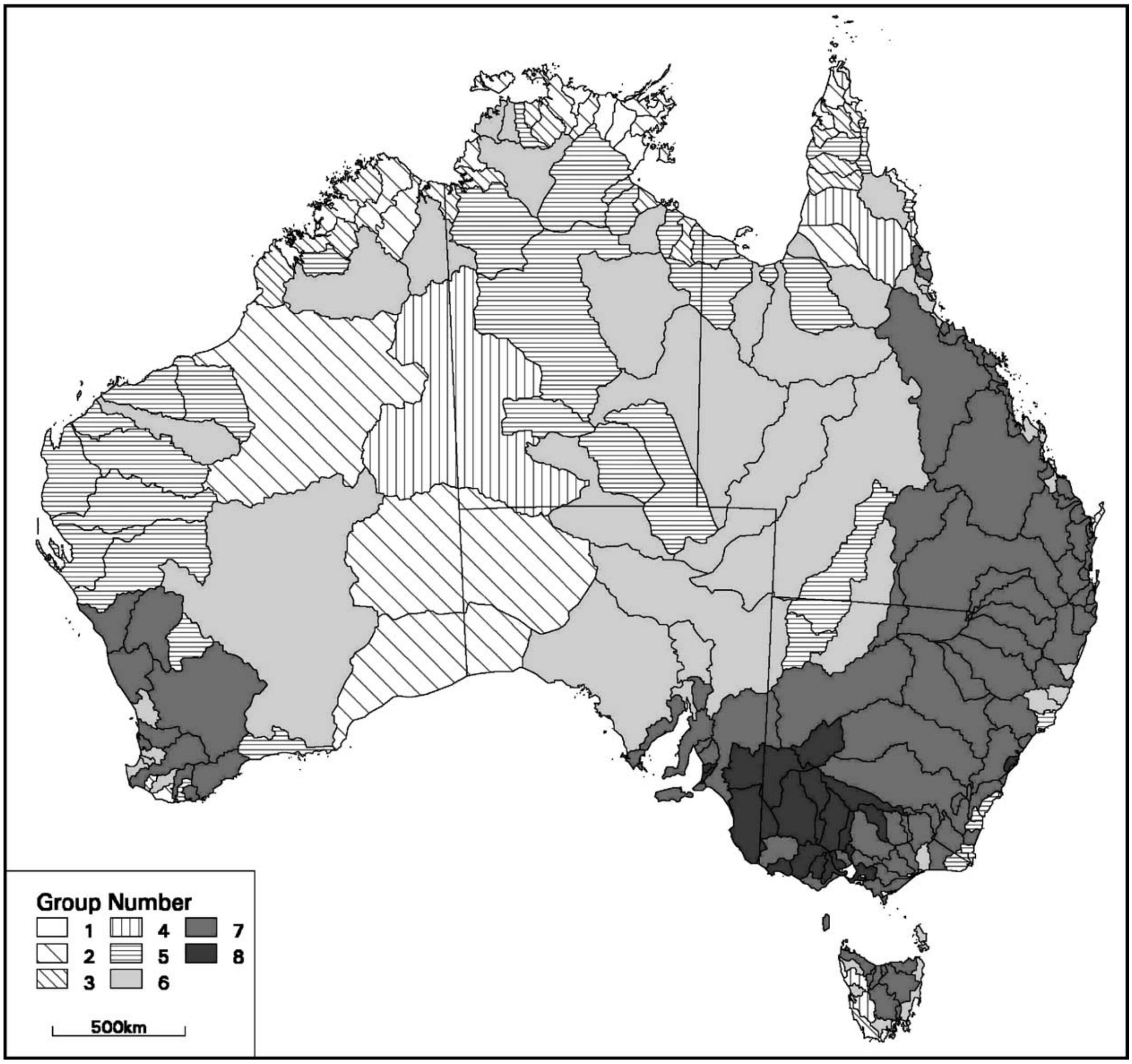

Fig. 7. River basins classified according to their RDI profile. Groups are ordered from least disturbed (group 1) to most disturbed (group 8).

retained so values for selected catchments or individual stream sections can be interrogated or mapped. Fig. 8 shows two examples for the Yarra River catchment: (1) demonstrating alternative thresholds for selecting potential wild rivers; and (2) identifying stream sections undisturbed by infrastructure development. This catchment, extending to the north and east of the city of Melbourne, covers an area of 409,562 ha and supplies the majority of the city's water supply (Department of Water Resources Victoria, 1989). Its lower reaches, flowing through the city, are highly degraded, yet in the forested, mountainous headwaters a number of streams remain undisturbed. Increasing the RDI threshold value for potential wild rivers from 0.01 to 0.05 increases the number of qualifying stream sections from 84 to

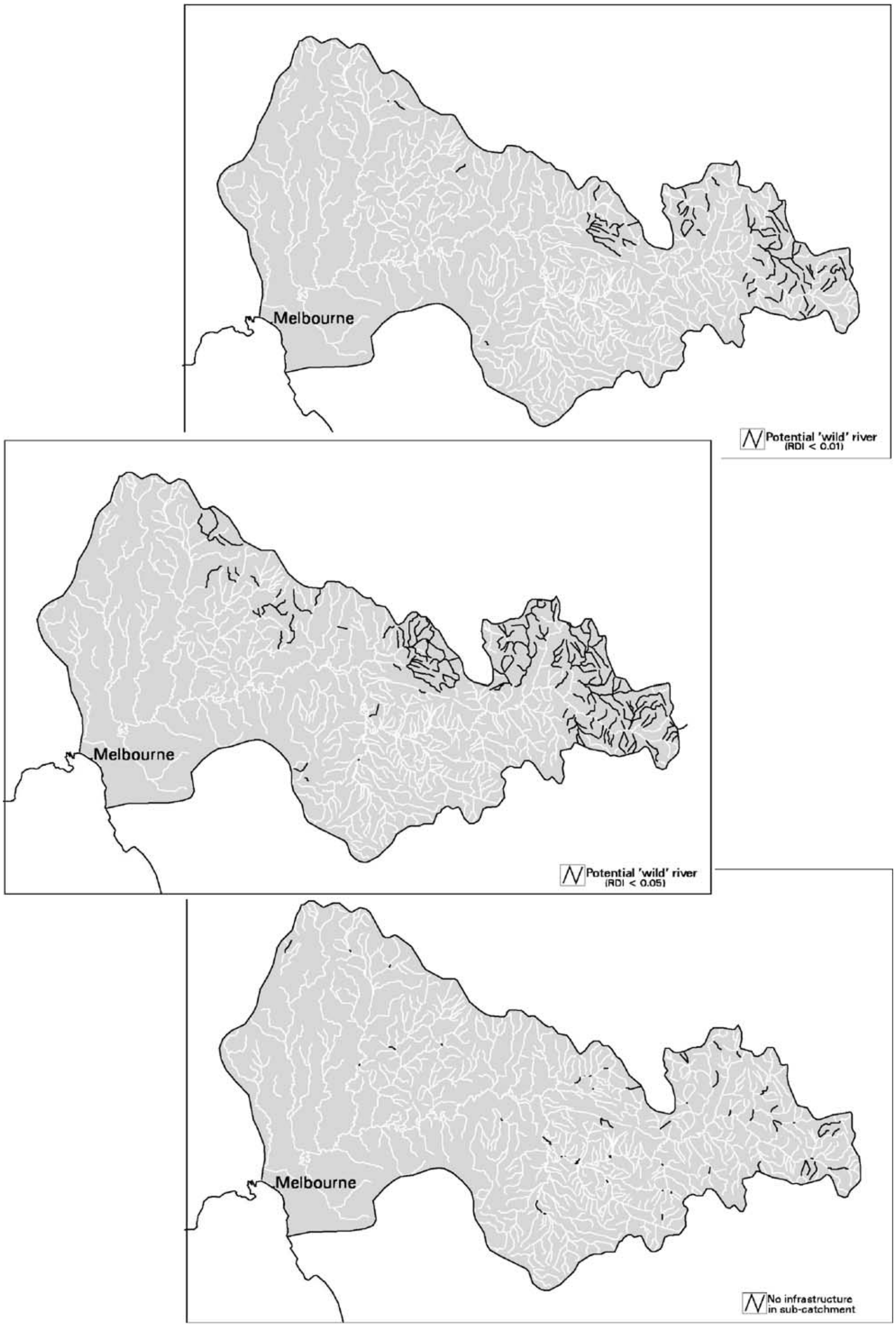

Fig. 8. Application of the river disturbance database: examples from the Yarra River catchment. Potential wild rivers may be identified as river sections with RDI values less than a specified threshold, for example, <0.01 (top) or <0.05 (middle). River sections from roadless sub-catchments may be located by identify stream sections with IF values equal to zero (bottom).

Table 7
Stream length by RDI class for river basin groups. RDI values shown are class upper limits.

Group	Proportion of stream length in RDI class (%)									Stream length (km)	Area (km^2)	Number of basins
	0.01	0.1	0.2	0.3	0.4	0.5	0.6	0.7	1			
1	95	5	0	0	0	0	0	0	0	23203	37959	6
2	79	20	1	<1	0	0	0	0	0	178888	1150231	21
3	47	50	2	0	0	0	0	0	0	96530	162589	18
4	50	45	4	<1	<1	<1	<1	0	0	128828	489623	6
5	17	61	21	<1	<1	0	0	0	0	586635	1395437	40
6	15	56	29	<1	<1	<1	<1	<1	0	1122671	2506615	45
7	5	27	51	12	2	1	1	2	<1	875564	1729374	88
8	<1	5	33	23	15	7	3	12	1	50078	221732	20
Total	19	45	30	4	<1	<1	<1	<1	<1	3063680	7693560	244

Table 8
Stream length by upstream catchment area class. RDI values shown are class upper limits

Catchment area class (km^2)	Proportion stream length in RDI class (%)									Stream length (km)
	0.01	0.1	0.2	0.3	0.4	0.5	0.6	0.7	1	
<100	21	44	30	4	<1	<1	<1	<1	<1	2642129
100–1000	11	47	32	4	2	1	1	1	<1	283526
1000–5000	6	50	27	5	2	1	3	5	1	82349
>5000	2	37	25	4	4	3	7	18	<1	56223
Total	19	45	30	4	<1	<1	<1	<1	<1	3063680

214. Field checks or additional information are required to determine the condition of each of these streams. Despite reservation of the Upper Yarra catchment for water supply in 1888 (Department of Water Resources Victoria, 1989) very few stream sections are free of infrastructure (Fig. 8). Because of the importance of roadless areas for biodiversity conservation (Strittholt and Dellasala, 2001) and the increasing rarity of roadless areas, especially roadless catchments, it is critical to maintain such areas in their natural state (Trombulak and Frissell, 2000). Such streams would also be a valuable source of baseline information for studies assessing the impact of roads on stream biota.

5. Discussion

A spatial method has been developed as a generic tool for mapping anthropogenic disturbance of river systems and thus, at the undisturbed end of a continuum, wild rivers. The necessity for national data coverage dictated the use of methods that were reliant on indirect indicators using available databases. This also had the advantage of being more accessible to, and interpretable by, a lay audience. The method can be applied at any scale, but is most applicable at scales ranging upwards from regional to continental.

The disturbance indices are indicators of pressure, as in the Pressure-State-Response (PSR) model framework presented by the OECD (OECD, 1998) and adopted for State of the Environment Reporting in Australia (State of Environment Advisory Council, 1996). The PSR model considers that human activities exert pressures on the environment and affect its quality and the quantity of natural resources (state). The link between the disturbance indices as indicators of pressure, and the state of a river could be examined by correlation with the many indicators of river health, but issues of scale, both temporal and spatial, may confound comparisons. Activities in the catchment inevitably influence the stream although this may take many decades to occur (Harding et al., 1998; Boulton and Brock, 1999). As a result the full effects of such activities may be difficult to detect (Findlay and Bourdages, 2000). Most appropriate for comparison will be those measures of river health that integrate long term changes at landscape scales. To date, validation of RDI scores has focused on the potential wild rivers, however, Harris and Silveira (1999) found broad concurrence between RDI values and an Index of Biotic Integrity (IBI) calculated for the fish communities at 80 sites covering a wide range of environmental condition across New South Wales.

The nation-wide characterization of river disturbance could potentially support many other river planning and management tasks including: (1) conservation planning, as an indicator of river naturalness (Dunn, 2000) or vulnerability (e.g. Pressey et al., 1996); (2) identification of the source(s) of observed river degradation; (3) prioritization of river restoration and rehabilitation (e.g. Rutherford et al., 2000); (4) development of systematic survey strategies for aquatic, riparian and estuarine biota; (5) testing of biotic indices at a series of sites of known anthropogenic disturbance (e.g. Burton et al., 1999); and (6) analysis of the extent of degradation, often underestimated by site specific studies (Hughes et al., 2000).

However, such applications must acknowledge the limitations of this assessment. It is likely that there will be significant variation in the response of streams to disturbance, both regionally and with river type. In this case, a standard set of weights was used, reflecting both the need for a consistent national overview and the absence of a detailed understanding of regional variations in river response to disturbance. Differences between upland and lowland reaches are partially accounted for by the weighting of sub-catchment disturbance scores by a surrogate for run-off. Most 'run-off' is generated in upland reaches so disturbance in these reaches is given greater weight in the overall catchment scores. An appropriate regionalization could be incorporated by setting regionally specific weights and distance thresholds. For example, a higher weighting for CDI may be warranted in upland rivers where dominance by allochthonous material means the catchment condition has a strong influence on the character and function of the aquatic system (Young et al., 2001). Of course the interpretation of disturbance index scores, for example, the threshold applied to identify potential wild rivers, can be varied regionally, recognizing that similar levels of disturbance need not equate with similar levels of degradation.

The disturbance indices are formulated simply as the linear weighted combination of component factors or indices. There are more sophisticated mathematical approaches that could be used in deriving disturbance indices, but those adopted here give greatest weight to simplicity, transparency and cognition by non-specialists.

Attributing the level of disturbance as some function of proximity to a disturbing factor satisfies common sense, but assigning a particular distance decay function is essentially arbitrary. Alternative functions could be substituted where appropriate. The contribution of a tributary to the condition of the main stream is assumed to be relative to its contribution to total stream run-off. In this way, the impacts of disturbance may be 'diluted' downstream by undisturbed tributary inflows. However, no decay of impact function is assumed. Clearly, for some disturbances, it would be appropriate to include a decay function to account for the behaviour of pollutants within the stream and associated floodplain wetlands. Such functions potentially require a large amount of input data to account for the many factors that influence this behavior (Capel et al., 2001). However, only a simple, generic decay function is likely to be feasible within the scope of a national assessment.

Operation of the method depends on the estimated surface flow paths derived from a DEM. Of necessity, these comprise a simplified representation of the way water moves across the landscape that is sensitive to errors and inadequacies in the DEM. A planned revision of the assessment of Australian streams will benefit from the recently completed major upgrade of the national DEM and its associated surface drainage structure (Geoscience Australia, 2001).

This upgrade entailed further extensive revision of the source elevation and streamline data and enhancement of the ANUDEM elevation gridding procedure.

Groundwater flow was not considered explicitly, yet groundwater inputs can provide a major component of the flow of many rivers, particularly in periods of low flow. While indirect indicators, for example, vegetation clearance, drainage and irrigated agriculture, potentially alter groundwater flows, the relationships were insufficiently known to be used in this study.

This national assessment of Australian streams highlighted limitations of the supporting data. For most of the indirect indicators only the occurrence of a feature is recorded in the database—there is no indication of its direct impact on a stream. Within any one category there may be large variation in the actual impact of an activity, depending on its management. For example, the location of an operating or abandoned mine is recorded, but the type of operation, waste management, etc. are unknown. The weights assigned to each category are essentially arbitrary but those used here were based on an understanding of the relative impact of different activities and were subjected to extensive review. They are used only to provide a relative ranking along the disturbance continuum and can be readily varied from the software interface. The categories chosen were partly dictated by the available data. Additional or different categories could also be used, in most cases without any changes to the software.

In addition to inadequate description of some features, much data of relevance to river condition were unavailable at national scale. These included: the condition of riparian vegetation; changes in fire frequency and intensity; the presence of exotic species; the intensity of catchment grazing by feral animals; and the location of river engineering works (e.g. channel straightening, desnagging). Where available such information could be readily accommodated within the existing method.

Drainage networks in arid zones differ markedly from those in humid areas, being characterized by much greater variability in discharge, long periods without flow and regimes dominated by flood discharges that move slowly down long, low gradient channels (Knighton and Nanson, 1994, 2001). In comparison with more humid areas, discharge in arid streams varies in a more complex way with drainage area, relationships changing from positive to negative because of increased transmission losses in the larger catchments (Knighton and Nanson, 2001). Disturbance in arid lands may have a very significant impact locally, for example, significant movement of sediment due to grazing pressure around a watering point. However, these effects may not be carried downstream as run-off decreases due to evaporation or infiltration into sandy beds. In such cases, the distance decay function might be more appropriately replaced by reference to whether the disturbance was on or off the floodplain. However, it was not possible to identify those areas within the floodplain from the 10–20 m elevation resolution available from a 1:100,000 or 1:250,000 scale DEM. Even with very much higher spatial resolution, such an assessment is problematic since elevation differences of centimeters can determine flows. The unique characteristics of arid drainage systems will require special consideration in future revisions of this assessment.

6. Conclusions

A method for assessing anthropogenic river disturbance at continental and regional scales is described. Its application provides the first national assessment of river disturbance in Australia and highlights the scarcity of undisturbed river systems. The database developed in this study can provide a basis for many river management and planning tasks, applications well beyond the initial objective of identification of wild rivers. It can also assist in interpretation of the causes of observed environmental degradation. However, the primary focus of its development has been the identification of the least disturbed, wild rivers. Further work is needed to quantify the impact of different types of disturbance, evaluate regional differences and to examine the correlations between river disturbance indices and intensively sampled physical and biological indicators of river condition.

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References

Ali, M.M., Sreekishnan, T.R., 2001. Aquatic toxicity from pulp and paper mill effluents: a review. Adv. Environ. Res. 5, 175–196.

AUSLIG, 1992. GEODATA TOPO-250K User Guide. Australian Survey and Land Information Group, Canberra.

AUSLIG, 1997a. GEODATA 9 Second DEM User Guide. Australian Survey and Land Information Group, Canberra, http://www.auslig.gov.au/products/digidat/dem9s/dem9s_ug.htm.

AUSLIG, 1997b. River Basins of Australia, Version 1.0. Australian Survey and Land Information Group, Canberra.

Bargagli, R., Monaci, F., Borghini, F., Bravi, F., Agnorelli, C., 2002. Mosses and lichens as biomonitors of trace metals. A comparison study on *Hypnum cupressiforme* and *Parmelia caperata* in a former mining district in Italy. Environ. Pollut. 116, 279–287.

Baxter, R.M., 1977. Environmental effects of dams and impoundments. Ann. Rev. Ecol. Syst. 8, 255–283.

Belbin, L., 1993. PATN, Version 3.6. CSIRO Division of Wildlife and Ecology, Canberra.

Boulton, A.J., 1999. An overview of river health assessment: philosophies, practice, problems and prognosis. Freshwater Biol. 41, 469–479.

Boulton, A.J., Brock, M.A., 1999. Australian Freshwater Ecology: Processes and Management. Gleneagles Publishing, Adelaide.

Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.

Brierley, G.J., Cohen, T., Fryirs, K., Brooks, A., 1999. Post-European changes to the fluvial geomorphology of Bega catchment, Australia: implications for river ecology. Freshwater Biol. 41, 839–848.

Burton, T.M., Uzarski, D.G., Gathman, J.P., Genet, J.A., Keas, B.E., Stricker, C.A., 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. Wetlands 19, 869–882.

Calace, N., Liberatori, A., Petronio, B.M., Pietroletti, M., 2001. Characteristics of different molecular weight fractions of organic matter in landfill leachate and their role in soil sorption. Environ. Pollut. 113, 331–339.

Capel, P.D., Larson, S.J., Winterstein, T.A., 2001. The behavior of 39 pesticides in surface waters as a function of scale. Hydrol. Process. 15, 1251–1269.

Chessman, B., Growns, I., Currey, J., Plunkett-Cole, N., 1999. Predicting diatom communities at the genus level for the rapid biological assessment of rivers. Freshwater Biol. 41, 317–331.

Costa-Cabral, M.C., Burges, S.J., 1994. Digital elevation model networks (DEMON): a model of flow over hill slopes for computation of contributing and dispersal areas. Water Resources Res. 30, 1681–1692.

Crabb, P., 1997. Impacts of anthropogenic activities, water use and consumption on water resources and flooding. Australia: State of the Environment Technical Paper Series (Inland Waters). Department of the Environment, Canberra.

Croke, J., Hairsine, P., Fogarty, P., 2001. Soil recovery from track construction and harvesting changes in surface infiltration, erosion and delivery rates with time. For. Ecol. Manage. 143, 3–12.

Davies, B.R., Snaddon, C.D., Wishart, M.J., Thoms, M.C., Meador, M., 2000. A biogeographical approach to interbasin water transfers: implications for river conservation. In: Boon, P.J., Davies, B.R., Petts, G.E. (Eds.), Global Perspectives on River Conservation: Science, Policy and Practice. Wiley, New York, pp. 431–444.

Department of Water Resources Victoria, 1989. Water Victoria: A Resource Handbook. VGPO, Melbourne.

Donnelly, T.H., Ford, P.W., McGregor, D., Allen, D., 1999. Anthropogenic changes to a billabong in New South Wales. 1. Lagoon evolution and phosphorous dynamics. Mar. Freshwater Res. 50, 689–698.

Downes, B.J., Hindell, J.S., Bond, N.R., 2000. What is in a site? Variation in lotic macroinvertebrate density and diversity in a spatially replicated experiment. Aust. Ecol. 25, 128–139.

Doyle, M.W., Harbor, J.M., Rich, C.F., Spacie, A., 2000. Examining the effects of urbanization on streams using indicators of geomorphic stability. Phys. Geogr. 21, 155–181.

Dunn, H., 2000. Identifying and Protecting Rivers of High Ecological Value. Land and Water Resources Research and Development Corporation, OP 01/00, Canberra.

ESRI, 1996. ArcInfo, Version 7.0.4. Environmental Systems Research Institute, Inc., Redlands.

Farrell, T.P., Kratzing, D.C., 1996. Environmental effects. In: Mulligan, D.R. (Ed.), Environmental Management in the Australian Minerals and Energy Industries: Principles and Practices. University of New South Wales Press, Sydney, pp. 14–45.

Findlay, C.S., Bourdages, J., 2000. Response time of wetland biodiversity to road construction on adjacent lands. Conserv. Biol. 14, 86–94.

Forman, R.T.T., 2000. Estimate of the area affected ecologically by the road system in the United States. Conserv. Biol. 14, 31–35.

Forman, R.T.T., Alexander, L.E., 1998. Roads and their major ecological effects. Ann. Rev. Ecol. Syst. 29, 207–231.

Franklin, J.F., 1992. Scientific basis for new perspectives in forests and streams. In: Naiman, R.J. (Ed.), Watershed Management. Balancing Sustainability and Environmental Change. Springer, New York, pp. 25–72.

Gardner, T., Geary, P., Gordon, I., 1997. Ecological sustainability and on-site effluent treatment systems. Aust. J. Environ. Manage. 4, 144–156.

Geoscience Australia, 2001. GEODATA 9 Second DEM, Version 2. http://www.auslig.gov.au/products/digidat/dem.htm.

Gibbins, C.N., Jeffries, M.J., Soulsby, C., 2000. Impacts of an inter-basin water transfer: distribution and abundance of *Micronecta poweri* (Insecta: Corixidae) in the River Wear northeast England. Aquat. Conserv. Mar. Freshwater Ecosyst. 10, 103–115.

Gilliom, R.J., 2001. Pesticides in the hydrologic system—what do we know and what is next? Hydrol. Process. 15, 3197–3201.

Harbor, J., 1999. Engineering geomorphology at the cutting edge of land disturbance: erosion and sediment control on construction sites. Geomorphology 31, 247–263.

Harding, J.S., Benfield, E.F., Bolstad, P.V., Helfman, G.S., Jones III, E.B.D., 1998. Stream biodiversity: the ghost of land use past. Proc. Natl. Acad. Sci. U.S.A. 95, 14843–14847.

Harding, J.S., Young, R.G., Hayes, J.W., Shearer, K.A., Stark, J.D., 1999. Changes in agricultural intensity and river health along a river continuum. Freshwater Biol. 42, 345–357.

Harris, J.H., Silveira, R., 1999. Large-scale assessments of river health using an index of biotic integrity with low-diversity fish communities. Freshwater Biol. 41, 235–252.

Hart, B.T., Maher, B., Lawrence, I., 1999. New generation water quality guidelines for ecosystem protection. Freshwater Biol. 41, 347–359.

Holmgren, P., 1994. Multiple flow direction algorithms for run-off modeling in grid-based elevation models: an empirical evaluation. Hydrol. Process. 8, 327–334.

Hughes, R.M., Paulsen, S.G., Stoddard, J.L., 2000. EMAP-surface waters: a multiassemblage, probability survey of ecological integrity in the USA. Hydrobiologia 422, 429–443.

Hutchinson, M.F., 1989. A new method for gridding elevation and streamline data with automatic removal of pits. J. Hydrol. 106, 211–232.

Hutchinson, M.F., 1996. ANUDEM, Version 4.4. Center for Resource and Environmental Studies, Australian National University, Canberra.

Jackson, J., 1997. State of habitat availability and quality in inland waters. Australia: State of the Environment Technical Paper Series (Inland Waters). Department of the Environment, Canberra.

Jager, H.I., Chandler, J.A., Lepla, K.B., Van Winkle, W., 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. Environ. Biol. Fish 60, 347–361.

Jansen, A., Robertson, A.I., 2001. Relationships between livestock management and the ecological condition of riparian habitats along an Australian floodplain river. J. Appl. Ecol. 38, 63–75.

Jenson, S.K., 1991. Applications of hydrologic information automatically extracted from digital elevation models. Hydrol. Process. 5, 31–44.

Jenson, S.K., Domingue, J.O., 1988. Extracting topographic structure from digital elevation data for geographic system analysis. Photogramm. Eng. Remote Sens. 54, 1593–1600.

Jesson, L., Kelly, D., Sparrow, A., 2000. The importance of dispersal, disturbance, and competition for exotic plant invasions in Arthur's Pass National Park, New Zealand. N. Z. J. Bot. 38, 451–468.

Jolly, I., Caitcheon, G., Donnelly, T., Hafner, S., 1996. Physical and chemical indicators of water quality. In: Walker, J., Reuter, D.J. (Eds.), Indicators of Catchment Health: A Technical Perspective. CSIRO, Melbourne, pp. 131–141.

Kelly, M., 1988. Mining and the Freshwater Environment. Elsevier, London.

Kelly, V.J., 2001. Influence of reservoirs on solute transport: a regional-scale approach. Hydrol. Process. 15, 1227–1249.

Kingsford, R.T., 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. Aust. Ecol. 25, 109–127.

Knighton, A.D., Nanson, G.C., 1994. Flow transmission along an arid zone anastomosing river, Cooper Creek, Australia. Hydrol. Process. 8, 137–154.

Knighton, A.D., Nanson, G.C., 2001. An event-based approach to the hydrology of arid zone rivers in the Channel Country of Australia. J. Hydrol. 254, 102–123.

Ladson, A.R., White, L.J., Doolan, J.A., Finlayson, B.L., Hart, B.T., Lake, P.S., Tilleard, J.W., 1999. Development and testing of an index of stream condition for waterway management in Australia. Freshwater Biol. 41, 453–468.

Lake, P.S., Palmer, M.A., Biro, P., Cole, J., Covich, A.P., Dahm, C., Gibert, J., Goedkoop, W., Martens, K., Verhoeven, J., 2000. Global change and the biodiversity of freshwater ecosystems: impacts on linkages between above-sediment and sediment biota. BioScience 50, 1099–1106.

Lane, P.N.J., Mackay, S.M., 2001. Stream flow response of mixed-species eucalyptus forests to patch cutting and thinning treatments. For. Ecol. Manage. 143, 131–142.

Lesslie, R., Maslen, M., 1995. National Wilderness Inventory Australia. Handbook of Procedures, Content and Usage. Australian Heritage Commission, Canberra, http://www.heritage.gov.au/anlr/code/pub.html.

Liston, P., Maher, W., 1997. Water quality for maintenance of aquatic ecosystems: appropriate indicators and analysis. Australia: State of the Environment Technical Paper Series (Inland Waters). Department of the Environment, Sport and Territories, Canberra.

Lytle, D.A., Peckarsky, B.L., 2001. Spatial and temporal impacts of a diesel fuel spill on stream invertebrates. Freshwater Biol. 46, 693–704.

Maddock, I., 1999. The importance of physical habitat assessment for evaluating river health. Freshwater Biol. 41, 373–391.

Maher, W., Batley, G.E., Lawrence, I., 1999. Assessing the health of sediment ecosystems: use of chemical measurements. Freshwater Biol. 41, 361–372.

Maingi, J.K., Marsh, S.E., 2002. Quantifying hydrologic impacts following dam construction along the Tana River, Kenya. J. Arid Environ. 50, 53–79.

Malanson, G.P., 1993. Riparian Landscapes. Cambridge University Press, Cambridge.

Marks, D.M., 1988. Network models in geomorphology. In: Anderson, M.G. (Ed.), Modeling Geomorphological Systems. Wiley, Chichester, pp. 73–98.

McMahon, J.P., 1995. ANUCLIM Users Guide, Version 1. Center for Resource and Environmental Studies, Australian National University, Canberra.

Moore, I.D., Grayson, R.B., Ladson, A.R., 1991. Digital terrain modeling: a review of hydrological, geomorphological and biological applications. Hydrol. Process. 5, 3–30.

Nisbet, T.R., 2001. The role of forest management in controlling diffuse pollution in UK forestry. For. Ecol. Manage. 143, 215–226.

Nix, H.A., 1981. Simplified simulation models based on specified minimum data sets: the CROPEVAL concept. In: Berg, A. (Ed.), Application of Remote Sensing to Agricultural Production Forecasting. Commission of the European Communities, Luxembourg, pp. 151–169.

OECD, 1998. Towards Sustainable Development: Environmental Indicators. Organization for Economic Cooperation and Development, Paris.

O'Neill, R.V., Hunsaker, C.T., Jones, K.B., Rütters, K.H., Wickham, J.D., Schwartz, P.M., Goodman, I.A., Jackson, B.L., Baillargeon, W.S., 1997. Monitoring environmental quality at the landscape scale. BioScience 47, 513–519.

Pen, L.J., 1999. Managing Our Rivers: A Guide to the Nature and Management of the Streams of Southwest Western Australia. Western Australia Water and Rivers Commission, East Perth.
Peters, D.L., Prowse, T.D., 2001. Regulation effects on the lower Peace River, Canada. Hydrol. Process. 15, 3181–3194.
Petts, G.E., 1984. Impounded Rivers: Perspectives for Ecological Management. Wiley, Chichester.
Pressey, R.L., Ferrier, S., Hager, T.C., Woods, C.A., Tully, S.L., Weinman, K.M., 1996. How well protected are the forests of north-eastern New South Wales? Analyses of forest environments in relation to formal protection measures, land tenure, and vulnerability to clearing. For. Ecol. Manage. 85, 311–333.
Rutherford, I.D., Jerie, K., Marsh, N., 2000. A Rehabilitation Manual for Australian Streams, Vol. 1. Land and Water Resources Research and Development Corporation and Cooperative Research Center for Catchment Hydrology, Canberra.
Sheldon, F., Thomas, M.C., Berry, O., Puckridge, J., 2000. Using disaster to prevent catastrophe: referencing the impacts of flow changes in large dryland rivers. Regul. River 16, 403–420.
Smith, M.J., Kay, W.R., Edward, D.H.D., Papas, P.J., Richardson, K.S., Simpson, J.C., Pinder, A.M., Cale, D.J., Horwitz, P.H.J., Davis, J.A., Yung, F.H., Norris, R.H., Halse, S.A., 1999. AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. Freshwater Biol. 41, 269–282.
Sokal, R.R., Michener, C.D., 1958. A statistical method for evaluating systematic relationships. Univ. Kansas Sci. Bull. 38, 1409–1438.
Stanford, J.A., Ward, J.V., 1992. Management of aquatic resources in large catchments: recognizing interactions between ecosystem connectivity and environmental disturbance. In: Naiman, R.J. (Ed.), Watershed Management: Balancing Sustainability and Environmental Change. Springer, New York, pp. 91–124.
State of Environment Advisory Council, 1996. Australia: State of the Environment 1996. An Independent Report to the Commonwealth for the Department of the Environment, Sport and Territories. CSIRO Publishing, Melbourne.
Stein, J.L., Stein, J.A., Nix, H.A., 1998. The Identification of Wild Rivers: Methodology and Database Development. Environ-ment Australia, Canberra, http://www.heritage.gov.au/anlr/code/pub.html.
Stein, J.L., Stein, J.A., Nix, H.A., 2001. Wild rivers in Australia. Int. J. Wilderness Rivers 7, 20–24.
Strittholt, J.R., Dellasala, D.A., 2001. Importance of roadless areas in biodiversity conservation in forested ecosystems: case study of the Klamath–Siskiyou ecoregion of the United States. Conserv. Biol. 15, 1742–1754.
Thoms, M.C., Ogden, R.W., Reid, M.A., 1999. Establishing the condition of lowland floodplain rivers: a palaeo-ecological approach. Freshwater Biol. 41, 407–423.
Townsend, C.R., Riley, R.H., 1999. Assessment of river health: accounting for perturbation pathways in physical and ecological space. Freshwater Biol. 41, 393–405.
Trombulak, S.C., Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conserv. Biol. 14, 18–30.
Vaughn, C.C., Taylor, C.M., 1999. Impoundments and the decline of freshwater mussels: a case study of an extinction gradient. Conserv. Biol. 13, 912–920.
Walker, K.F., Thoms, M.C., Sheldon, F., 1992. Effects of weirs on the littoral environment of the River Murray, South Australia. In: Boon, P.J., Calow, P., Petts, G.E. (Eds.), River Conservation and Management. Wiley, Chichester, pp. 271–292.
Watanabe, N.C., Harada, S., Komai, Y., 2000. Long-term recovery from mine drainage disturbance of a macroinvertebrate community in the Ichi-kawa River, Japan. Hydrobiologia 429, 171–180.
Williams, P.J., Pen, L.J., Stein, J.L., Prince, B., 1996. The identification, conservation and management of wild rivers in the Kimberley region of Western Australia. In: Stream Management'96 Proceedings of the First Australian Stream Management Conference. Cooperative Research Center for Catchment Hydrology, Merrijig, Victoria, pp. 157–162.
Young, W.J., Hillman, T.J., 2001. The Murray River—to the Darling River junction. In: Young, W.J. (Ed.), Rivers as Ecological Systems: The Murray–Darling Basin. Murray–Darling Basin Commission, Canberra, pp. 101–118.
Young, W.J., Schiller, C.B., Harris, J.H., Roberts, J., Hillman, T.J., 2001. River flow, processes, habitats and river life. In: Young, W.J. (Ed.), Rivers as Ecological Systems: The Murray–Darling Basin. Murray–Darling Basin Commission, Canberra, pp. 45–100.