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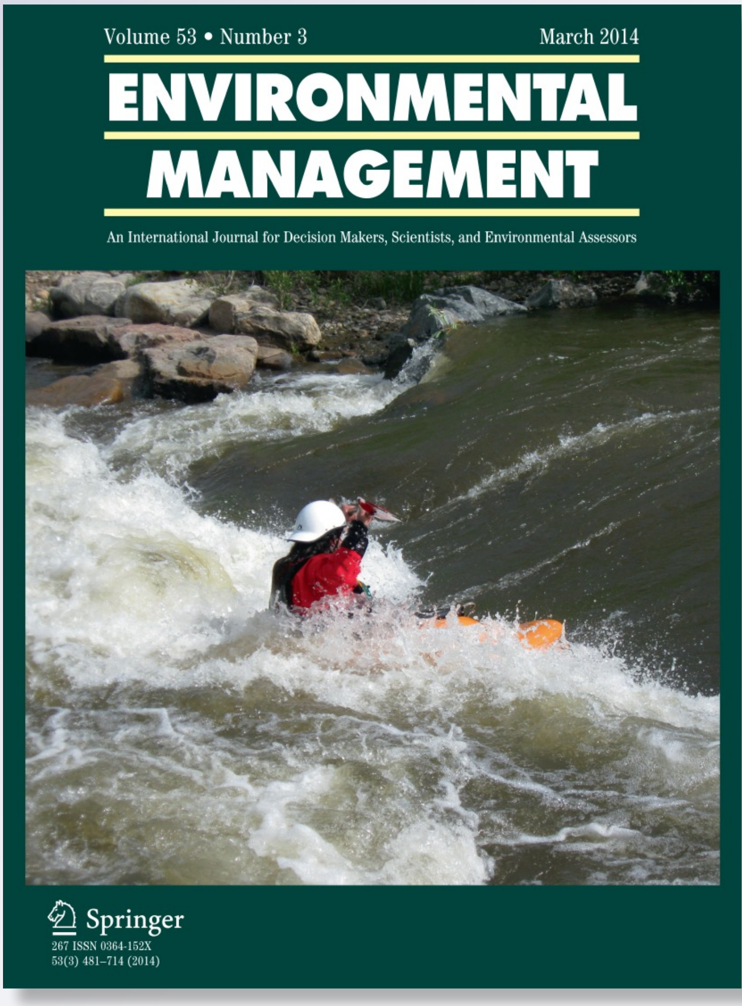

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Evaluation of Current Approaches to Stream Classification and a Heuristic Guide to Developing Classifications of Integrated Aquatic Networks

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Abstract Conservation and management of fresh flowing waters involves evaluating and managing effects of cumulative impacts on the aquatic environment from disturbances such as: land use change, point and nonpoint source pollution, the creation of dams and reservoirs, mining, and fishing. To assess effects of these changes on associated biotic communities it is necessary to monitor and report on the status of lotic ecosystems. A variety of stream classification methods are available to assist with these tasks, and such methods attempt to provide a systematic approach to modeling and understanding complex aquatic systems at various spatial and temporal scales. Of the vast number of approaches that exist, it is useful to group them into three main types. The first involves modeling longitudinal species turnover patterns within large drainage basins and relating these patterns to environmental predictors collected at reach and upstream catchment scales; the second uses regionalized hierarchical classification to create multi-scale, spatially homogenous aquatic ecoregions by grouping adjacent catchments together based on environmental similarities; and the third approach groups sites together on the basis of similarities in their environmental conditions both within and between catchments, independent of their geographic location. We review the literature with a focus on more recent classifications to examine the strengths and weaknesses of the different approaches. We identify gaps or problems with the current approaches, and we propose an eight-step heuristic process that may assist with development of more flexible and integrated aquatic classifications based on the current understanding, network thinking, and theoretical underpinnings.

Keywords Ecosystem · Conservation · Freshwater · Running water · Rivers · Typologies · Directional networks · Lakes · Wetlands · Large-scale ecology

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Introduction

Aquatic ecosystem classifications (AECs) help governments manage and conserve aquatic resources at regional, provincial, and national scales in the absence of complete data on every lake, river, or wetland (Soranno et al. 2010). A variety of approaches to AEC can be employed (e.g., Evans and Noble 1979; Snelder and Biggs 2002; Higgins et al. 2005), and the utility of these different approaches increases if they are based on the predictive relationships and a mechanistic (or physically based) understanding of how key ecosystem characteristics develop and respond to change (e.g., Goodwin 1999; Pyne et al. 2007; Soranno et al. 2010). Classification frameworks can serve a number of management goals such as: assessing ecosystem status, aquatics parks planning, conserving biota and their habitats, quantifying how ecosystems might respond to stressors, setting policy, extrapolating from data-rich areas to data-poor areas, and selecting reference sites (Hawkins et al. 2000; Soranno et al. 2010). Managers and conservation planners must keep in mind that classifications are hypotheses that should be iteratively tested and refined. A single classification will not likely suit all management goals (Loveland and Merchant 2004).

Freshwater aquatic ecosystems include surface waters such as streams, wetlands, and lakes together with reservoirs that are typically connected by flowing water and fed by precipitation and groundwater inputs. Despite a general recognition that lakes, wetlands, and rivers are integrally connected (Jones 2010; Soranno et al. 2010), development of classifications that explicitly incorporate effects of different freshwater ecosystem types on each other seems to be mired with difficulty. Much of the classification research to date has focused on each type of aquatic system in isolation (Soranno et al. 2010), but a few river classifications make attempts to integrate the influence of lakes. For example, Snelder et al. (2004b) developed a lake index, which takes both lake surface area and the volume of water passing through a lake in relation to total volume coming from the lake catchment into account. Here we focus on the design of integrated flowing water classifications that incorporate the influence of lakes and wetlands on streams and rivers. An explicit evaluation of lake (or wetland) classifications per se was beyond the scope of our research.

Several challenges must be addressed when creating integrated stream classifications: (1) determine the best way to divide branched networks of rivers, lakes, reservoirs, and wetlands into manageable and meaningful ecological units appropriate for measurement, comparison, and interpretation (Brenden et al. 2008a; Steel et al. 2010). (2) Determine the appropriate spatial scales [e.g., local reach contributing areas (RCA) and upstream catchments areas] and temporal scales (e.g., seasonal, yearly, historic averages, and post-glacial) for data collection and analysis (Fausch et al. 2002). Processes operating over small extents and fine grains evidently change more frequently than processes operating over large extents and coarse grains (Frissell et al. 1986), though small-scale disturbances can have system-level impacts in directional systems. (3) Determine how controlling factors such as climate and geology affect key physical characteristics and how their influence changes with network position (Pyne et al. 2007; Finn et al. 2011) and spatial scale (e.g., Frissell et al. 1986; Fausch et al. 2002). (4) Determine how the modifying effects of lakes, rivers, and wetland processes in different landscape positions can be incorporated (Kratz et al. 1997; Soranno et al. 1999). (5) Determine how vertical and lateral processes can be incorporated (i.e., groundwater and floodplain interactions with the stream or water body, Ward 1989, e.g., Borwick et al. 2006).

Perhaps due in part to the challenges faced when it comes to modeling and understanding stream ecosystems (Melles et al. 2012), aquatic classification has fallen behind terrestrial classification (Turak and Linke 2011; Hermoso et al. 2011). Although many governments have made regional, national, and international commitments to protect freshwater ecosystems (Stein and Nevill 2011), the scientific and policy frameworks for their protection are still poorly developed (Hermoso et al. 2011; Nel et al. 2011). Freshwater systems often garner inadvertent protection as political borders or features within protected areas designed to support terrestrial ecosystem classes and land-based species of interest (Hermoso et al., 2011); but a terrestrial reserve system will likely fail to capture even the most basic elements of aquatic representation, such as stream size (Stein and Nevill 2011). Protecting representative stream ecosystem types is not as straightforward as protecting land-based systems because these systems are directionally connected, branching networks (Nel et al. 2011; Turak et al. 2011; Heiner et al. 2011). Yet without regional, national, and international frameworks for aquatic classification, there is a limited ability to deal with the complexity and diversity of aquatic ecosystems and to assess whether or not management goals are being met.

Numerous stream classification approaches exist, not all of which are mutually exclusive (reviewed by Hudson et al. 1992; Naiman et al. 1992; Naiman 1998; Goodwin 1999; Gordon et al. 2004), and many of which are grounded in theoretical developments that have influenced and transformed our understanding of flowing waters over the last century (Melles et al. 2012). Three common approaches are: longitudinal zonation (Hawkes 1975), regionalized hierarchical classification (Frissell et al. 1986), and place-independent approaches (Rosgen 1994). Longitudinal zonation approaches emphasize physical and biotic conditions that describe the replacement or addition of species from headwaters to outlets (e.g., Huet 1954, 1959; Illies 1961; McGarvey and Hughes 2008). Regionalized (or place-dependent) hierarchical approaches group freshwater systems into spatially contiguous ecoregions at a variety of spatial scales based on the observed patterns of variation in controlling processes such as geology and climate (Frissell et al. 1986; Maxwell et al. 1995). Regionalized approaches pay less regard to differences that may occur in different positions of a watershed in the creation of spatially contiguous ecoregions (e.g., Higgins et al. 2005; Abell et al. 2008). Place-independent approaches can group freshwater systems into successive categories according to similarities in their physical and biotic factors, without the constraint of spatial contiguity. For example, rivers may be grouped on the basis of their gradient or slope, degree of entrenchment, and dominant bed material (e.g., sand, gravel, and bedrock, Rosgen 1994), or rivers may be classified according to their biotic community structure (Wright et al. 1998; Noble et al. 2007; Poquet et al. 2009). But streams of a given type in place-independent classifications can be found anywhere within the classification domain because they tend to place less emphasis on spatial contiguity and/or regional geoclimatic differences (e.g., Rosgen 1994).

Many stream classifications include elements of all the three approaches. Place-independent approaches do not preclude consideration of upstream catchment conditions (e.g., Snelder and Biggs 2002; Seelbach et al. 2006; Brenden et al. 2008b), and they can also consider controlling factors at a variety of spatial scales (e.g., Snelder and Biggs 2002; Snelder et al. 2004a, 2005). Upper levels of regionalized hierarchical classifications are often partitioned from the top-down to form increasingly smaller nested units (Omernik 1987; Kleynhans et al. 2005; Higgins et al. 2005; Seelbach et al. 1997, 2006; Thorp et al. 2008). Lower level classes at finer spatial scales are then created through an iterative process involving top-down and bottom-up approaches. In bottom-up approaches, more objective statistical clustering or regression techniques are typically used to form groupings (Snelder et al. 2007), occasionally under spatial contiguity constraints (Brenden et al. 2008a; Snelder et al. 2010). The result of all these approaches is a mixture of regionalized, longitudinal, and place-independent classes at a variety of spatial scales. Indeed we could not find any examples in the literature of multiple-scale stream classifications created entirely from the bottom-up.

The goal of this paper is to review the predominant approaches to stream classification with a focus on more recent publications (i.e., since approximately 1980). Though several excellent reviews of stream classification methods exist (Hawkes 1975; Hudson et al. 1992; Naiman et al. 1992; Naiman 1998; Goodwin 1999; Gordon et al. 2004; Melles et al. 2012), there are no guidelines that provide a synthesis of the different methods given the current thinking in aquatic ecology. Each of the three main approaches to stream classification that we identify has associated strengths and weaknesses. One objective of this paper is to explore and contrast these approaches and highlight the importance of incorporating network thinking (as per Benda et al. 2004a) and a combined fluvial landscape mosaic approach (as per Poole 2002) based on sound ecological theory (Melles et al. 2012).

In a recent review, Melles et al. (2012) examined how theoretical developments over the last century in geomorphology and stream ecology have influenced stream classification and conservation planning, but they did not assess many of the methodological issues with stream classification, nor did they provide specific recommendations about how to go about developing one. Here we extend their evaluation of the recent literature to provide guidelines for classification of stream ecosystems. We identify gaps or problems with the current approaches that may explain why many fluvial classifications fail to be fully adopted (Pennak 1971; Hudson et al. 1992; Soranno et al. 2010). With this evaluation in mind, we propose an outline for the classification of directionally nested aquatic networks at multiple spatial scales.

Methods: Evaluating the State of the Literature

Our main focus was on classification of flowing waters (i.e., river or lotic classifications) because research in this area has gone further toward integrating and understanding how lake, wetland, and groundwater influences change riverine environments (Ward and Stanford 1983; Peterson et al. 2009; Jones 2010). We searched Web of Science databases in December 2010 for articles with the keywords "stream, river, aquatic, fluvial, or floodplain" and "classification" in the title. We then followed up on relevant references noted in key works, some of which were not in modern databases. Many classifications are applied science and the product of government and non-government agencies that are not found in the primary literature. We searched for gray literature using Google search engine, government libraries, and coauthor personal libraries of government reports. A total of 81 classifications were compiled for review (Appendix 1 in Supplementary Material). The majority of studies were from Canada (12) and the United States (30) with good representation from Europe (14) and global systems (13); a few studies were from New Zealand (5), South Africa (3), Australia (2), Korea (1), and China (1). The studies we reviewed were published predominantly after 1990.

In this review, we first determined the magnitude and number of spatial scales of ecosystem analysis. We further recorded whether groupings made at each scale were longitudinal, place-dependent (regionalized), place-independent, or some combination. We noted whether classifications focused on catchment- or gradient-based (longitudinal) measures, and we noted whether measures of network position and connectivity were included. We determined whether groupings were found using aggregative bottom-up techniques, or divisive top-down approaches. Where specified, criteria for variable selection were recorded, and the purpose of the classification was also recorded.

A variety of explanatory variables are used in different approaches to stream classification; these variables may be grouped into three categories: abiotic, biotic, or a combination of both, whereby abiotic variables are typically used as predictors of biotic classes (sensu Brenden et al. 2008b). Abiotic-based classifications incorporate physical and chemical characteristics including numerous parameters that attempt to describe the physico-chemical characteristics of different species' habitat at a variety of spatial scales. There are at least four different subcategories of abiotic variables: geoclimatic, geomorphic, hydrologic, and chemical (Gordon et al. 2004). In contrast, biotic-based classifications group observational units predominantly on the basis of the presence of various taxa, such as: game fish (Shelford 1911; Ricker 1934; Huet 1959; Illies 1961; Anonymous 1967), benthic invertebrates (Knight and Gaufin 1967; Wright et al. 1984; Biggs et al. 1990), aquatic macrophytes (e.g., used by Pennak 1971 and others), and riparian vegetation; or some combination of biota (e.g., Rivers-Moore and Goodman 2010).

All the reported classification variables were sorted into these types: abiotic, biotic, or their combination; and we further subdivided abiotic variables into geoclimatic, geomorphic, hydrologic, or chemical (as per Portt et al. 1989; Gordon et al. 2004).

1. Geoclimatic-based classification systems use climate (temperature and precipitation regimes), physiography, and surficial geology as the fundamental basis for classification. These classifications are based on the premise that streams draining areas with similar climate and geology will have similar biotic conditions because climate and geology have a controlling influence on dependent variables such as soils, vegetation, and water (Lotspeich and Platts 1982), which help characterize species' habitat. Climate influences water temperature and flow regimes, which affect the distribution of many species (Gaston 2003). Macro- and meso-climate interact with geology to produce landforms (Lotspeich and Platts 1982; Portt et al. 1989), which can further limit species distributions; and geology influences groundwater quantity, which moderates stream water temperatures (Arrigoni et al. 2008).
2. Geomorphic-based classifications take a structural engineering approach to classification and are widely used at both the reach (Rosgen 1994; Montgomery and Buffington 1997) and valley-segment scales (Frissell et al. 1986; Maxwell et al. 1995; Brierley and Fryirs 2000). The basic premise of geomorphic classifications is that channel morphology is governed by the laws of physics through fluvial processes and readily observable features of stream channels (e.g., valley wall and channel slope and substrate characteristics, Rosgen 1994; Montgomery and Buffington 1997). Channel morphology is then related to specific habitat characteristics that are associated with different (fish) species (Montgomery and Buffington 1998).
3. Hydrologic-based classifications use numerous indices of flow to classify streams, including measures of flow variability such as: the timing, duration, magnitude, frequency, and rate of change in flow. These indices also summarize whether a stream is ephemeral, intermittent, or permanent. Additional measures can include mean daily, seasonal, or annual flow, groundwater levels, baseflow, and water source of flow (Kim and Hawkins 1993; Poff and Ward 1989; Zhang et al. 2011). Many authors consider flow to be a "master variable" because it is strongly correlated with many abiotic and biotic characteristics of interest in stream classification (Poff et al. 1997; Gordon et al. 2004).
4. Chemical or water quality-based classifications are often designed to assess the physical and chemical characteristics in streams according to criteria established by various environmental protection agencies (e.g., Ahl 1994; Dodds et al. 1998; Gordon et al. 2004). The criteria of interest often attempt to capture the degree of eutrophication as reflected by nutrient levels such as nitrogen and phosphorus, and toxicity of stream pollutants that negatively affect human and ecosystem health (e.g., pH and toxins).

Biotic-based classifications were subdivided into four groups: fish, benthic invertebrates, aquatic macrophytes, and measures of riparian vegetation.

Some clarification around use of terms like reach, segment, and catchment is necessary. Although these words have intuitive meaning that is linked to their relative size and application, they are often used interchangeably. In our review, there was a lack of consensus around use of the terms reach and segment in particular. Reaches are generally defined as small stretches of streams (<1 km) between (for example) bends, pools, and riffles (Frissell et al. 1986; Bisson et al. 2007). However, in practice, reaches typically refer to confluence to confluence links in a spatial database (Geographic Information System, GIS), and a RCA is the drainage area that contributes water flow to that section of stream (e.g., Peterson and Ver Hoef 2014). Frissell et al. (1986) define segments as confluence to confluence sections of stream flowing through a single bedrock type, which can be broken by waterfalls or other significant geomorphic features. However, many authors describe "valley" segments as portions of the network that are uniform in their hydrogeomorphology (1–10 km), which often break at stream junctions, but can contain multiple spatially adjacent interconfluence sections of stream (e.g., Maxwell et al. 1995; Benda et al. 2004b; Bisson et al. 2007). As such, valley segments are expected to have relatively homogeneous biological communities, and are advocated as appropriate spatial units for monitoring and management (Maxwell et al. 1995; Seelbach et al. 2006; Brenden et al. 2008a). In the interest of clarity, we adopted the practical (GIS) definition of reach as a confluence to confluence section of stream, and the hydrogeomorphic definition of segment as a homogeneous valley segment that may contain multiple reaches.

Results and Evaluation

The majority of classifications we examined were multi-level classifications that included summaries at various spatial scales (Appendix 2 in Supplementary Material). Almost half took a regionalized approach. Of the classifications with more than one level, two-thirds were regionalized (i.e., place-dependent) top-down and expert-driven approaches for several spatial scales. A combination of top-down and bottom-up approaches was used in almost one quarter cases, such that finer scales were classified in a place-independent manner, whereas coarser scales were classified into spatially contiguous ecoregions using expert-driven approaches. Place-independent approaches were used in all but one single-level classification (Appendix 2 in Supplementary Material). Some place-independent, multilevel classifications restricted clustering of aquatic units within broader aquatic ecoregions (e.g., Higgins et al. 2005). Whereas longitudinal, gradient-based approaches were used in one third of the studies we examined. Longitudinal zonation approaches to classification were not distinguished in our analysis from approaches that used gradient-based or regression-like models to describe aspects of the aquatic environment. This group also included difficulty to place classifications that used methods like spatial autocorrelation analysis (e.g., Thompson et al. 2006).

Research on the effects of confluences and tributaries on stream characteristics is building (Benda et al. 2004a; Rice et al. 2008), but there is less research on how lakes and wetlands might modify stream characteristics and vice versa (Jones 2010). Less than one third of the stream classifications we reviewed included lakes in their classification (Appendix 1 in Supplementary Material). Of these classifications, even fewer explicitly characterized lakes; rather, lakes were examined in terms of how they modified the riverine environment. Furthermore, none of the studies we reviewed integrated wetlands to create a cohesive aquatic classification.

Two of the most commonly stated reasons for developing a stream ecosystem classification were ecosystem management, and inventory and monitoring (Fig. 1). Ecosystem classification helps characterize key environmental processes and species' physical habitat requirements, and they provide an aid to responsible resource use planning when the goal is to manage ecosystem processes and maintain freshwater species distributions across broad landscapes. Classifications also provide a stratification framework for freshwater inventory and monitoring programs that feed state of the environment reporting across broad aquatic ecoregions. Protection of representative examples of unique aquatic ecosystem types in parks is recognized as a critical goal (Fig. 1) in the face of declines in biodiversity and increasing resource development (Leathwick et al. 2010).

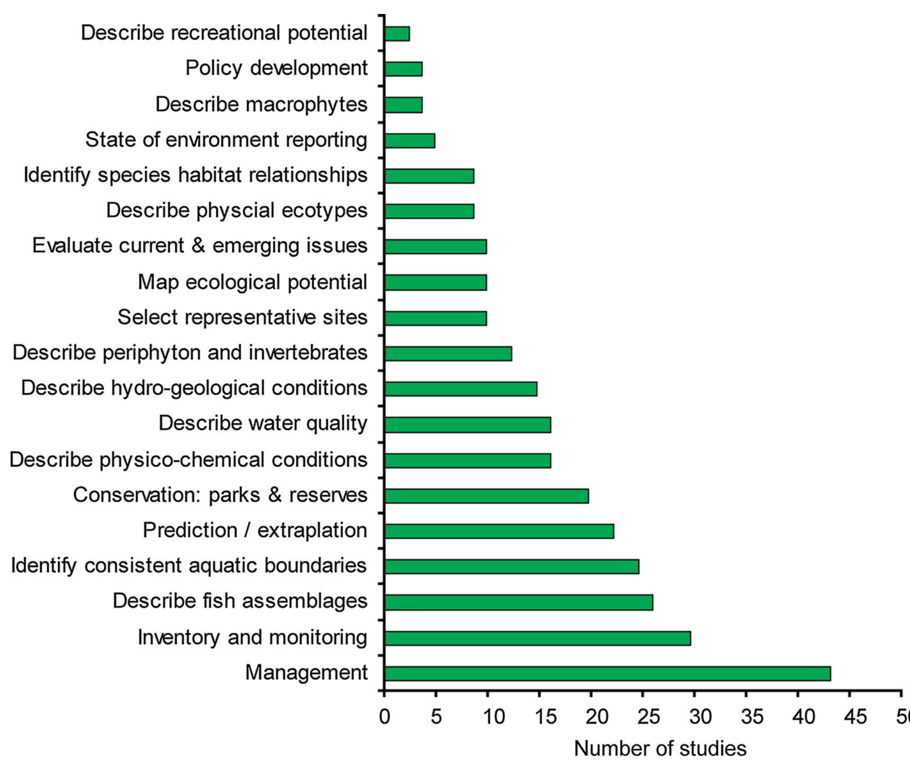

Fig. 1 Number of classifications with the stated purpose(s), based on our review of $n = 81$ classifications

Selection of Dominant Variables and Spatial Scales

In the past, many European and North American classifications were based largely on dominant game fish species assemblages arranged longitudinally into community zones (see Melles et al. 2012 for a more thorough treatment of this topic, e.g., Carpenter 1927; Anonymous 1967). Though game fish-based classifications are still prevalent in classifications developed since 1990, abiotic and more diverse biotic variables are more prevalent in the classifications of today (Wright et al. 1984, 1989, 2000; Fig. 2). For example, just under half of the classifications we reviewed used data on fish assemblages to inform aquatic classes (Fig. 2), and even fewer studies stated that a description of fish assemblages was a key goal (Fig. 1). This suggests that focus is shifting toward a more diversified picture of aquatic ecosystems that includes environmental descriptions relevant to other taxonomic groups (e.g., an abiotic habitat template approach that examines environmental gradients to which all species respond; Southwood 1977; Huryn et al. 2005; Jones et al. 2010). Most authors realize that both abiotic and biotic factors contribute to community structure at various times (Schoener 1987; Resh et al. 1988; Poff and Ward 1989; Death 2010). Determining the relative importance of abiotic and biotic factors in time and space for individual species in the community is one of the most challenging problems of ecology today.

Variables used to classify aquatic units were measured at a variety of spatial scales including: stream reach, river segment or segment catchments, whole system (lake, wetland, river, and estuary), catchment, watershed, and ecoregion (Fig. 3). The intermediate "segment level" scale was the focal unit size for most classifications (Fig. 3) as this unit size seems to represent a trade-off between the need to cover broad spatial extents and the need for detailed aquatic information. Variables measured at the level of river reaches can be assessed at greater levels of detail and finer spatio-temporal scales (e.g., current speeds, dominant bed materials, and river width:depth ratios) than variables summarized at watershed levels (e.g., physiography and geologic history). Yet it would be unrealistic to measure or map microhabitat variables for all the river reaches and aquatic features of interest across the entire classification domain. Though remote sensing technologies are available to map the whole river systems at fine spatial scales, these technologies remain too expensive to use over very broad spatial extents, and intensive field work is still necessary to collect many physical and biotic variables.

Of the studies that listed criteria for variable selection, the majority selected variables that covaried with key ecological factors and controlling processes (Fig. 4). Of course, data availability was also a principle concern for classification models intended to cover large spatial extents. The availability of information that was easily quantified was identified as an important consideration in almost half of the studies we examined (Fig. 4). The five most prevalent abiotic factors measured or modeled were geology, hydrology, morphology, topography, and climate.

Fig. 3 Key variables and aquatic spatial units used in AEC ($n = 81$).▶ Each block represents one classification. A reach in these studies is defined as a small stretch of stream between bends, pools, and riffles; a segment is defined as either a section of stream between confluences, or a segment includes several confluence to confluence sections of stream; whole system refers to entire lakes, wetlands, or rivers; a watershed is a drainage area contributing precipitation and groundwater to a particular river; an ecoregion is a relatively homogeneous, ecologically distinctive land area defined by a variety of factors such as geology, landform, soil, climate, water, and biota

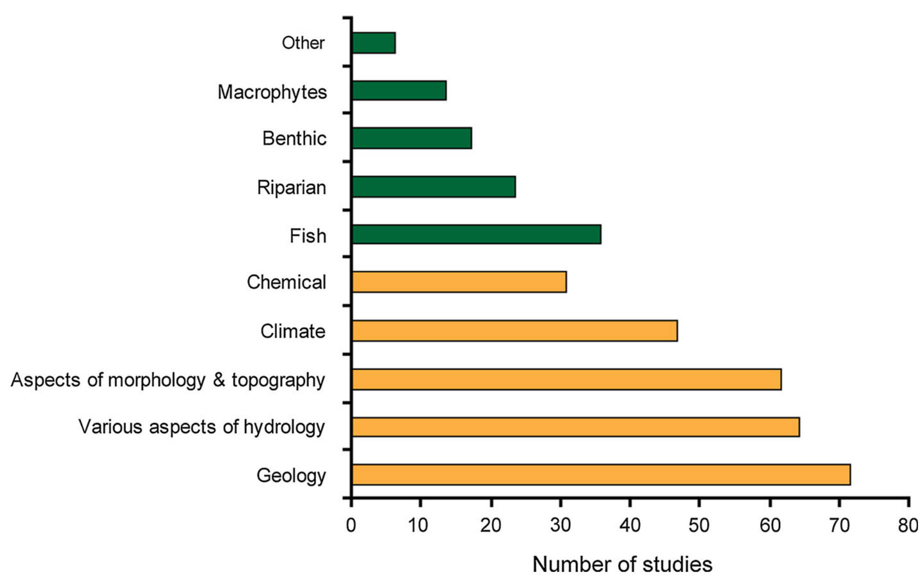

Fig. 2 Number of studies that used these classes of dominant variables in their classifications ($n = 81$). Biotic variables are shown in *dark green* and abiotic variables are shown in *light orange*


REACH
Streamflow
Sediment/substrate
Width, depth, entrenchment
Channel & valley shape
Slope
Sinuosity
Bank, channel stability
Turbidity, grain size
Flora & fauna
Pools, runs, riffles
Riparian vegetation
Water chemistry
Stream order, size, position
Deposition pattern
Debris occurrence
Stability
Soil / rock type
Temperature
Habitat
Temporality
Water source
Altitude of site
Bed roughness
Bank composition
Longitudinal profile
Channel migration
WHOLE SYSTEM
(Lakes, wetlands, rivers, estuaries)
Chemistry & turbidity
Dominant plant spp, PNV
Streamflow
Trophic state
Substrate type
Sediment transport
Riparian vegetation
Elevation/gradient, location, size class
Temperature
Area
Particle size
Presence of dams / levees
Macrophytes
inflows, lake order, network struct.
Shoreline exposure
Fish species
SEGMENT
Geology: rock & soil type, lithology, geologic permeability, sediment
Gradient (slope), valley slope, topography, relief
Stream size, stream order, upstream catchment area
Temperature (e.g., mean July, mean annual, winter, latitude)
Flow (changes, discharge class, disturbance regime, runoff)
Chemistry (nutrients, chlorophyll a, turbidity, pH, conductivity)
Elevation
Precipitation (e.g., mean annual)
Channel confinement, aspect
Landcover
Drainage network position
Fish species
Vegetation type, instream habitat
Substrate, suspended sediment
Lake depth, channel depth
Lake size, lake influence ratios
Lake/basin/channel morphology
Drainage density, pattern metrics
Groundwater linkage
Species composition (aqu/riparian)
Benthic invertebrates
Potential natural vegetation
Water source (glacier, lake, peat)
Connectivity (up/downstream)
Stream width
Local evaporation line
Length (longest) watercourse
Stable isotopes
Vertical/lateral floodplain activity
Presence of dams/levees
WATERSHED
Soil type & vegetation
Physiography
Fish community
Watershed area
Slope
Landuse
% carbonated materials
% evaporitic materials
Maximum altitude
Drainage divide
Fish distribution
Cross section
Geologic history
Rock type
Glacial refuges
Migration barriers, historic connections
Annual discharge
Watershed biogeoclimatic class
ECOREGION
Climate measures
Bedrock geology
Topo relief, landform
Zoogeog. patterns
Surficial geology
Vegetation patterns
Elevational gradient
Soils
Glacial history
Ecoregion class
Hydrography
Rainfall
Paleo-geologic origin
Land use
PNV
Fluvial evolution pttrn.

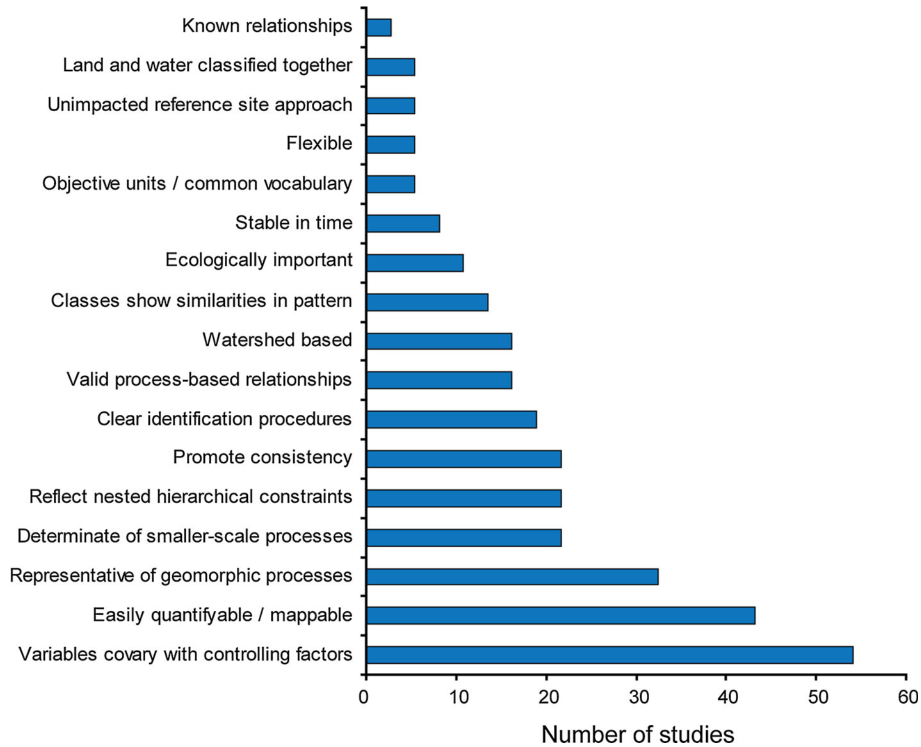

Fig. 4 Number of classification papers that listed criteria for variable selection ($n = 37$)

Some measure of geology (substrate or soil type) was included in the majority of aquatic classifications. Surficial and bedrock geology are widely recognized as having the greatest biological significance to aquatic communities (Pennak 1971). Geology affects groundwater quantity via baseflow (Piggott and Sharpe 2007; Power et al.1999; Poole 2010), which thereby influences water temperature (Arrigoni et al. 2008) and streamflow stability (Poff and Ward 1989). Geology also affects water chemistry (Rimet 2009; Neff and Jackson 2011), topography, physiography, surface water hydrology, and channel morphology (Leopold and Maddock 1953; Peterson et al. 2009). These interactions form a "hydrogeomorphological" basis for stream ecology (Poole 2010). Among the reviewed studies hydrologic and morphologic measures were more common than climatic measures such as 30-years climate norms.

Chemically based or water quality measures were less commonly used in the classifications we reviewed. These measures tend to be used more often in Europe and in more heavily populated areas where legislation exists to protect water quality from human degradation and pollutants. Such classifications tend to relate biotic community patterns to physical/chemical conditions (Naiman et al. 1992; Wright et al. 1998), and deviation from reference conditions is used to determine site quality. However, under the Water Framework Directive (Council of the European Communities 2000), there is a push to use more ecologically based measures such as geology, hydrology, river continuity, and morphological conditions (Hatton-Ellis 2008).

Variables that account for the position of an aquatic feature within a watershed (e.g., stream order and drainage area; Fig. 3) were included at a variety of spatial scales. Their attraction is likely because these variables have a profound influence on stream ecology: they encapsulate a variety of hydrologic and geomorphic processes, which then affect biotic community structure (Peterson et al. 2009; Poole 2010). The use of stream order or basin area in a classification, however, was not necessarily based on a complete understanding of how stream environmental heterogeneity changes with these variables (Hughes et al. 2011). If anything, their indiscriminate use in a classification implements a log linear view of aquatic networks (i.e., the River Continuum Concept or RCC, Vannote et al. 1980), a concept that has been largely criticized (Poole 2010; Melles et al. 2012). The RCC does not adequately consider network effects, the relative size of tributaries, resultant confluence ratios (Benda et al. 2004a), or lake effects (Jones 2010). Apart from stream order and drainage area, other measures of network position (i.e., distance to outlet and number of connected lakes up- or down-stream) were rarely examined as important considerations in the lotic classifications we reviewed. Perhaps it is time to move beyond stream order and drainage area? Olden et al. (2010) suggested that more robust quantifications of stream network topology (i.e., metrics that capture the geometric properties and connectivity of stream networks) are needed to explore observed regional and local species diversity patterns and longitudinal species-discharge relationships in aquatic environments. In the sections that follow, we further discuss the need to incorporate network thinking in stream classifications as well as other gaps in our understanding; but first we discuss the strengths and weaknesses of the three main approaches to stream classification.

Strengths and Weaknesses of the Three Main Approaches

Longitudinal Zonation

Longitudinal zonation has considerable relevance to cataloguing freshwater diversity because species richness increases with increases in mean annual discharge along with concomitant changes in species composition. The predictive species-discharge and distance-decay relationships similar to land-based species–area curves can be used to model expected changes in diversity with river regulation (McGarvey 2010; Olden et al. 2010). With increases in mean annual discharge there are also predictive changes in channel size and drainage area. Indeed, the physical structure of aquatic networks dictates that larger mid- and higher-order systems behave differently from the smaller headwater systems that they drain largely because of differences in their size, flow, position, and inputs (Fullerton et al. 2010; Finn et al. 2011). The long-supported RCC (Vannote et al. 1980), which is now viewed as an oversimplification (Poole 2010, Melles et al. 2012), provided a model for classifying river zones into indicator species or functional feeding groups based on the observed longitudinal changes in the physical characteristics and organic matter inputs of rivers. Headwater streams tend to be occupied by different suites of biota, such as colonizing specialists and species with the ability to withstand more unstable environmental conditions, than downstream systems (Vannote et al. 1980; McGarvey 2010; Finn et al. 2011). In turn, it makes ecological sense to categorize a river into faunal zones. The method is appealing as it provides a whole system approach to classification that can model changes across large basins (McGarvey and Milton-Ward 2008).

Disadvantages to longitudinal zonation approaches include the cost of collecting detailed field data to adequately describe the biota of entire river basins. As a result, zonation approaches typically focus on a single taxonomic group (e.g., fish or macroinvertebrates), neglecting the other aspects of biodiversity and possibly oversimplifying relationships between discharge and snapshots of species richness patterns (Fausch et al. 2002). Historic and seasonal influences tend to be ignored though this is a problem for most ecosystem classifications. Longitudinal approaches that look at biota and physical or chemical aspects tend to focus on the internal characteristics of discrete sections of stream. Thus, longitudinal classifications may not adequately account for regional geoclimatic differences (Hawkes 1975; Omernik and Bailey 1997).

Additionally, longitudinal zones may be separated by very long transitions and there may be no evident demarcations between biotic communities because many species move by varying amounts (e.g., salmonids vs. sculpin vs. invertebrates) and use different habitat types at different times of year. Rather gradual transitions in species associations are interrupted by marked changes below significant tributaries (Hawkes 1975; Bruns et al. 1984), and small-scale interruptions in longitudinal patterns complicate the analysis (Hawkes 1975; Poole 2002). Nevertheless, all the ecological classifications create artificial divisions and simplified classes along continuous environmental gradients.

Regionalized Hierarchical Classifications

A regionalized hierarchical structure for classification has several strengths (Frissell et al. 1986): classification at a coarse spatial scale sets the context and narrows the set of variables required at finer scales; the approach allows for integration of diverse sets of available data with various resolutions; and it allows land-use planners, managers, and researchers to select a classification scale that is most appropriate for their application. Lower level (i.e., finer scale) aquatic units are considered as integrated products of the watersheds surrounding them: these classifications assume that streams in similar geoclimatic regions will have similar characteristics; and if they are located within the same zoogeographic region, they will likely also have similar biotic communities. They provide a means for extrapolating what stream characteristics would be like at locations where field data are unavailable, but where regional abiotic patterns are known. Data-rich systems of a class can be used to extrapolate to data-poor systems within the same class.

One obvious limitation to regionalized, place-dependent classifications is that they pay less regard to the longitudinal or directional nature of aquatic networks discussed above. Though streams are generally described as natural hierarchical systems in which climate, geology, and topography at large spatial scales set the context for geomorphic processes that maintain heterogeneity at smaller scales (Frissell et al. 1986; Fausch et al. 2002), the way in which these controlling factors influence stream characteristics is position dependent in aquatic networks and this potentially violates the assumptions of hierarchy theory (Melles et al. 2012). Headwater streams are smaller; they integrate the prevailing conditions over smaller areas and are subject to more drying and spates than larger systems (Fisher 1997). Pyne et al. (2007) commented that it would be meaningless to compare environmental degradation in a second-order catchment in the Big Horn Mountains of Wyoming to the entire Mississippi River drainage. Yet many regionalized aquatic classifications inherently compare headwater stream segments with main stem segments within the same ecoregion or aquatic drainage unit because clustering or grouping is done without taking position within the watershed into consideration (e.g., Maxwell et al. 1995; Wolock et al. 2004; Higgins et al. 2005; Abell et al. 2008). One could have two river segments with similar physical properties that are geographically nearby, but located in different branches of the same network, and their biotic communities may differ substantially because they are located downstream from different types of headwaters (Pyne et al. 2007). To avoid this problem, some authors have classified headwater streams separately (Pyne et al. 2007; Finn et al. 2011).

Though the regionalized approach has intuitive appeal because it creates a simplified picture of aquatic types that can then be mapped, managed, and reported upon, there is little evidence to suggest that aquatic systems are more fittingly grouped into spatially homogeneous ecoregions than place-independent groups with non-contiguous boundaries (Jenerette et al. 2002; Wolock et al. 2004). Broad ecoregions cannot account for fine-scale (within watershed) variability in temperature, topography, geology, and land cover (Snelder et al. 2004a; Pyne et al. 2007; Chu and Jones 2010). Although fine-scale phenomena are nested within the coarse-scale regionalizations, research has shown that the position of aquatic features within a watershed affects the way local systems (e.g., lakes) respond to broad-scale changes (Kratz et al. 1997). For instance, Kratz et al. (1997) showed that network position influences how lakes respond to drought conditions with lakes on the outer branches of the network responding to a 4-year drought with decreases in calcium mass and lakes low in the landscape responding with increases in calcium mass. Not only do these findings have implications for species whose distributions are limited by calcium, but they also imply that *not* all lakes within a given climatic zone will respond similarly to climatic changes, which is a basic tenet of regionalized hierarchical classification (Bailey 1983).

As with all the classifications, there is a danger of oversimplification when gradual changes in aquatic characteristics are divided or lumped into ecosystem classes. Setting boundaries and class breaks for ecosystem mapping often involves many subjective, expert-driven decisions, and it is difficult to assess the uncertainties associated with these decisions (but see Chan et al. 2012). This is likely one of the key reasons why many streams ecosystem classification fail to be fully adopted.

Place-Independent Approaches

Place-independent approaches cluster units together on the basis of similarities in environmental or biological attributes, and they do so independent of their geographic location. This approach may be more ecologically relevant than regionalized classifications and results in an improved ability to discriminate variation in biological patterns at moderate to high levels of classification detail (Leathwick et al. 2011). Leathwick et al. (2011) reason that place-independent approaches capture and compare variation in the local environment (i.e., of stream segments) more effectively than regionalized approaches that emphasize factors varying over much broader spatial scales. Additionally, place-independent approaches can use newly emerging, more rigorous, and potentially more objective classification procedures such as generalized dissimilarity modeling (Ferrier 2002), which can discriminate parallel sets of biological data (i.e., data from multiple taxonomic groups, Snelder et al. 2007).

One of the drawbacks of place-independent approaches is that resultant typologies may have high spatial complexity, which could hinder explanatory power and lead to difficulties in interpretation of classes created from multivariate clustering algorithms that may themselves be difficult to grasp. Even when individual key process-driving variables and relationships are known and used in classification, it may be difficult to understand and interpret underlying causal linkages between resultant classes and these variables when they are used in combination. Results may seem weakly connected to underlying stream theory, and numerous small stream classes will likely be difficult to manage in a conservation context at regional scales. Additionally, most place-independent approaches do not use objective methods to create ecological aquatic units (as per Brenden et al. 2008a).

Identified Gaps in Our Understanding

Current research in stream ecology suggests that tributary streams modify the river continuum and the magnitude of adjustment depends on tributary size relative to the main stem as well as the physical, chemical, and biological characteristics of the tributary (Bruns et al. 1984; Rice et al. 2001; Benda et al. 2004a, b). These network relationships should be given consideration in modern aquatic classifications. Establishing useful aquatic network metrics that summarize ecological patterns in meaningful ways is a major knowledge gap requiring additional research (Grant et al. 2007; Eros et al. 2011).

Another gap in our understanding relates to how to integrate lakes and wetlands along with significant confluences and groundwater sources. Less than one third of the lotic and aquatic classifications we found explicitly included the effects of lakes. Yet we know that river networks characteristically include a variety of lakes, impoundments, wetlands, and groundwater inflows/outflows in different positions within the network, and these features are known to modify downstream conditions (Kratz et al. 1997; Riera et al. 2000; Martin and Soranno 2006; Jones 2010). Currently lakes and wetlands are typically represented as separate polygon layers in a GIS, and rivers are represented as poly-line networks. A major step toward developing integrated aquatic classifications involves technically representing both streams and other waterbodies together in a single spatial database that explicitly incorporates inlets, outlets, and flow direction.

There are also challenges in moving from fine to coarse spatial scales in aquatic environments (Stanfield 2012). Understanding how information is transferred when moving from one spatial scale to the next is important for creating multi-scale stream classifications. In land-based systems, moving to a coarser resolution and summarizing information over larger land units is a simple matter of resampling, and smoothing or aggregating information, for example using (weighted) mean, majority, mode, or median values. In aquatic systems, river networks may be "pruned" to coarsen the network, for example by removing the smallest (low order) streams, using a coarser resolution digital elevation model, or using algorithms that remove detail from convoluted lines. Yet it is not entirely clear how best to up- or down-scale information in aquatic networks (Stanfield 2012). It is not only difficult to determine which variables affect the characteristics of a given stream segment and how much of the upstream catchment area has an influence, but also difficult to then summarize information from all the stream segments in a network in a statistically valid way that adequately represents the character of larger drainage areas. Clearly, issues of scaling up are difficult to address in aquatic systems. In addition to the lack of readily available fine-scale data to aggregate over watershed scales, the problem of scaling may be why there are no examples of stream classifications created entirely from the bottom-up.

Aquatic ecosystem classes created using clustering approaches are generally based on data summaries from overlapping upstream catchment areas. Indeed, the similarity of stream catchments should increase as you move downstream as the area of overlap in upstream catchment areas increases. Therefore, any clustering within large drainage areas will likely find more similarities between 7th and 8th order streams than between 1st and 2nd order streams, all else being equal. Determining how best to address these types of directionally nested datasets over broad spatial extents seems to be another gap in our understanding. Recent geostatistical developments for streams account for spatial correlation, real stream distances, and downstream directionality, making detailed (regression-based) prediction of environmental variables possible (e.g., for water quality or stream temperature measures, Ver Hoef et al. 2006; Ver Hoef and Peterson 2010; Garreta et al. 2009). However, these methods cannot be implemented over broad spatial scales; they apply to the prediction of single variables, not the prediction of compound aquatic ecosystem classes.

Finding ways to incorporate uncertainty into classification models is another research need and knowledge gap. Recent applications of Bayesian mixture modeling (Webb et al. 2007; Achcar et al. 2009) that explicitly incorporate uncertainty in flow regime cluster outcomes (Kennard et al. 2010) are a promising new avenue to explore. Finding ways to estimate uncertainty of class membership (e.g., Chan et al. 2012) is necessary in any classification exercise but is rarely done in practice. Yet, if freshwater classifications fail to adequately capture uncertainty at regional, national, and international scales this could have major management implications for the design of aquatic reserves. For example, an uneven subdivision of total environmental variation could lead to over- or under-representation of these characteristics in aquatic reserve networks (Mackey et al. 1988; Snelder et al. 2004a).

We perceive the need to conceptualize a new theoretical framework for directional aquatic networks that combines knowledge of longitudinal variation in species turnover with knowledge that aquatic communities change in relation to regional variation at multiple spatial scales. Species turnover in rivers occurs both longitudinally, from headwaters to the river mouth (Muneepeerakul et al. 2008), and with regional environmental patterns or gradients (Higgins et al. 2005; Abell et al. 2008). Classification methods should be able to adequately represent both types of patterns, giving each an appropriate weight and representing species–environment relationships in a way that is consistent with underlying statistical and theoretical assumptions (Snelder et al. 2007; Leathwick et al. 2011).

Considerations for Classifying Directionally Nested Aquatic Networks

In this section, we describe an outline for classifying streams or more specifically directionally nested aquatic networks. But a note of caution is worthwhile here: any outline that we put forward is intended as a heuristic guide to stimulate thought and further research, rather than "a cage to confine it" (Hawkes 1975).

We propose a process that involves eight considerations (Fig. 5).

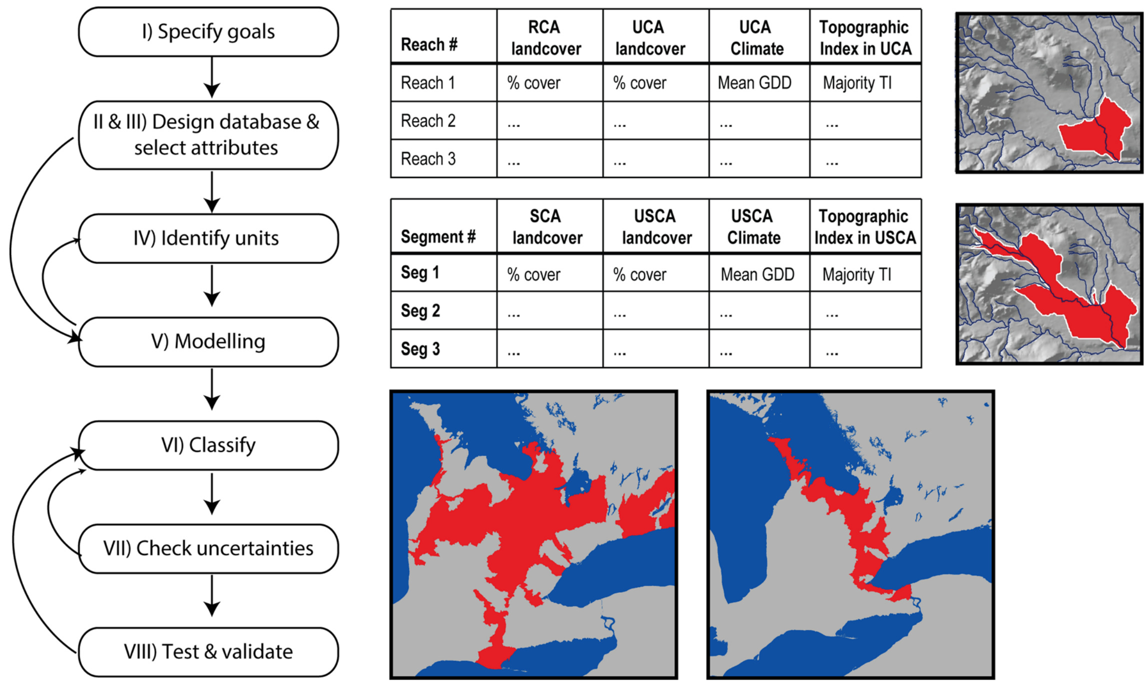

Fig. 5 Flowchart of eight considerations involved in the creation of classifications for integrated aquatic networks, including an example of database elements and illustrations showing their spatial equivalents. The two images of aquatic ecoregions illustrate the potential for creating different classifications using the same underlying data. *RCA* reach contributing area, *UCA* upstream contributing area, *SCA* segment contributing area, *USCA* upstream segment contributing area

(I) *Specify goals*. Even if multiple goals will be served by a variety of classifications (e.g., at different spatial scales), it is important to clearly state these goals up front. What is the primary purpose and how will the classification be applied? This will influence variable selection and database design (considerations II and III)

(II) *Design database*. Modern classifications need to remain flexible enough to accommodate emerging conceptual models, policy frameworks, and changing temporal dynamics (Melles et al. 2012). We recommend a database system and spatial framework that allows sections of stream to be attributed, queried, synthesized, and reported at a variety of spatial scales (i.e., within their RCAs, and upstream contributing areas, or UCAs, as per Wang et al. 2011, Fig. 5)

(III) *Select the smallest aquatic unit and attribute data layers*. The most basic spatial unit in the database needs to have both geomorphic and practical relevance in a GIS and associated hydrologic tools. We recommend that confluence to confluence reaches be used because they represent natural break points (i.e., nodes) in the type of line networks typically used in GIS-based stream network analysis. Spatially explicit up- and down-stream linkages are typically built into the data structure of these directional networks (Wang et al. 2011). Each *interconfluence reach* is coded with information about its position within the network (i.e., the *to* and *from* nodes based on direction of flow), allowing network analyses. Each record in the database would represent a single reach with fields summarizing attribute information. We also recommend explicitly including virtual lake and wetland connector line segments in the database by subdividing interconfluence reaches further at inlets and outlets of lakes and wetlands. The virtual connectors enable hydrologic analysis and modeling by allowing otherwise disconnected stream sections (separated by lakes and wetlands) to remain flow connected. Moreover, these virtual lake and wetland connectors can be attributed with associated data as well, essentially capturing the landscape character of the areas of land draining directly to the shorelines of lakes and wetland edges. A selection of attribute information (e.g., geology, land cover, climate, modeled regime variables, associated uncertainties, risk factors, and anthropogenic disturbances) can be summarized for each interconfluence reach to capture variation at both the local catchment (i.e., RCA) and upstream catchment scales (UCA)

Whether or not the entire up (or down) stream catchment scales need to be included when estimating key aquatic variables or attributes of interest will depend on underlying process relationships, statistical assumptions, and the goals of the classification. As discussed above, the word scale and the issue of changing scales in aquatic environments is problematic because "scale" changes as you move downstream: streams get larger and become rivers, and upstream contributing area increases as you move downstream. In our recent review of theoretical developments that underpin stream classifications (Melles et al. 2012), we point out that rivers are not spatially nested in the same way as land-based systems, and unlike land-based systems, rivers do not necessarily satisfy the assumptions of scaling according to hierarchy theory. River networks are directionally nested such that the main stem inherits properties from headwater streams, but headwaters do not inherit properties from distant main stem sections in the same way. Moreover, depending on the process of interest, not all upstream contributing areas will be relevant to calculation of a given attribute, but this will depend on prior knowledge and management goals.

(IV) *Select variables for classification*. The fourth matter is selection of variables that meet the goals of the classification (consideration I), either from the attribute database described above (consideration III), or derivative attribute data based on the predictive models. Ostensibly the most widely used and accepted abiotic variables for aquatic classifications (i.e., measures of geology, flow, topography, stream morphology, stream order/basin area/gradient, network positional metrics, and climate) are employed because they are related to four main underlying process regimes critical to aquatic ecosystem functioning and diversity. These four process regimes are (1) the flow regime (Poff et al. 1997; Poff and Zimmerman 2010), (2) the temperature regime (Fry 1971; Wehrly et al. 2006), (3) the nutrient, and (4) the sediment regime (Ryder 1965; Welcomme et al. 1989). These regimes affect the availability of different abiotic habitat types for all aquatic species and taxonomic groups because they affect food availability and organic matter, water clarity, water level, shelter, disturbance and survival, growing conditions, and breeding sites (as per Southwood 1977). The predictive models of these process regimes can be created using base attribute data, along with the appropriate uncertainty analyses so that users are aware of known measurement and model errors. Models may be complex (e.g., Arnold et al. 1993) or relatively simple (e.g., Piggott and Sharpe 2007). These different process regime models are derivatives of the original attribute database, and, although process regimes are important factors to consider, there are still other aspects or models of interest that can be generated from an attributed aquatic database (hence the flexibility of this approach)

(V) *Identify ecological units*. The basic inventory of interconfluence reaches (consideration III) represents no more than an unorganized list. For classification, it is important to determine ecological units (i.e., groupings of neighboring interconfluence reaches and virtual reaches) that are homogeneous (e.g., in their key process regimes, Seelbach et al. 1997; Brenden et al. 2008a, b). Resultant *stream valley segments* are expected to capture changes in many physical and biological processes that control habitat for a variety of species (Seelbach et al. 1997). Identification of homogeneous ecological units for classification is one of the most important and challenging aspects of ecosystem classification (Omernik and Bailey 1997; Loveland and Merchant 2004; Brenden et al. 2008a). Although expert-driven approaches to create significant valley segment boundaries were used in the past (e.g., Seelbach et al. 2006), there is a movement toward using more repeatable and objective approaches (e.g., Brenden et al. 2008a). Brenden et al. (2008b) developed a useful tool to group interconfluence sections of streams together based on a similarity criteria and spatial contiguity constraints

We posit that significant segments may be identified based on network metrics like confluence symmetry ratios (Leopold et al. 1964; Benda et al. 2004b). Moreover, we suggest that features such as the modifying effects of lakes, wetlands, groundwater, and floodplain processes may be incorporated during identification of these stream segments using variables that capture the zone or scale of influence of each process (Jones 2010; Ellis and Jones 2013). Stream valley segments and associated data (Figure 5) are useful for assessing, monitoring, and managing aquatic systems (Brenden et al. 2008a), and average conditions of river valley segments are likely to remain constant over the decade to century time scales. Depending on classification goals, temporal dynamics can be incorporated using predictor variables (e.g., using future climate model data), or by updating the classification database at regular intervals, and comparing results through time. Of course, the ecological relevance of the resultant stream segments should be validated (consideration VIII) according to the ecological phenomena of interest and study goals (consideration I) using field surveys (as per Warrner et al. 2010) and extensive observation or experimentation.

(VI) *Scaling up from valley segments*. Now that we have built stream valley segments and decided on the variables of interest based on management goals, the fifth step is the creation of objective class types. This step allows for multi-purpose, multi-scale classifications. Clustering can be performed in a variety of ways to suite user needs (e.g., by *K* means and various clustering algorithms, spatially constrained classification, classification and regression trees, maximum likelihood classification, and Bayesian mixture modeling). These clustering algorithms may be based on similarities in attribute data, modeled process regimes, or some combination, conducted over various scale domains (i.e., spatially constrained to watershed boundaries or regions). We foresee a range of classifications, examples *a* through *c* (described below), which represent the kinds of classifications we believe would be of interest depending on management goals

Classification a	To evaluate regional differences, headwater segments with basin areas below a threshold size may be clustered separately from downstream segments. This would insure that headwater segments are not classified together with segments draining significantly larger areas. Streams of third order or less make up nearly 85 % of all stream kilometers (Leopold et al. 1964). Some researchers argue that these headwater segments are likely to support a relatively impoverished biota (Milner et al. 2001), whereas others argue that they contribute substantially to regional scale biodiversity (Finn et al. 2011). We suggest that these are potential reasons to classify, test, and target unique headwater classes separately for resource planning purposes in order to reduce sampling efforts and minimize the risk of losing critical biodiversity
Classification b	Similarly, larger streams can be clustered separately as well, and the influence of headwater streams on downstream sections can be incorporated by including variables collected or modeled within upstream catchment areas (but see Ver Hoef et al. 2006; Ver Hoef and Peterson 2010; Garreta et al. 2009 for more detailed methods to make predictions about within-stream conditions). In addition, human activities such as urbanization, aggregate extraction, hydropower, and channel modifications tend to be situated near the outlet of a watershed, so these areas pose more challenges for management and conservation (Pringle 1997), and it is important to objectively identify their differences. Classifications *a* and *b* are functionally similar because clustering is restricted to catchments of a given size
Classification c	To evaluate up- and down-stream linkages and longitudinal patterns within the whole system, it is necessary to examine headwater, mid- and higher-order segments within the same watershed drainage area or aquatic zone. Watershed boundaries are important politically for making land and water management decisions (Blomquist and Schlager 2005). Spatially constrained clustering techniques may be tested for their ability to create larger and larger ecological units within a watershed by averaging information out over larger areas

(VII) *Evaluate of risks and uncertainties*. An examination of key risk factors and uncertainties affecting different ecosystem classes is an important consideration when using a classification because there are costs associated with being wrong. Risks include potential economic or lost opportunity costs associated with, for example, protecting areas that do not need protection because they were misclassified. In addition, there are potential environmental costs associated with classification error and uncertainty, such as losing species or species' habitat in areas that we fail to identify and manage adequately because they were misclassified. As suggested earlier, finding ways to incorporate uncertainty into classification is another research need and knowledge gap

(VIII) *Perform validation and testing*. The last and final necessary consideration in any AEC is to test the hypothesis that these groupings reflect actual differences on the ground. This step is related to an evaluation of risks and uncertainties (consideration VII). Classifications represent hypotheses about how we think ecosystems behave (Sokal 1974; Hawkins and Norris 2000). They must be put to the test to verify, dismiss, or alter proposed ecosystem classes. Do aquatic ecosystem classes actually reflect differences in species distributions and diversity (Van Sickle and Hughes 2000; Hawkins et al. 2000)? Does habitat for a variety of species vary in different segments more than within a given segment, and does our initial segment classification reflect the systems' natural ecological variation (Warrner et al. 2010)? Do the final ecosystem classes have ecological relevance? Feminella (2000) provides an example of a broadscale assessment done to explicitly examine the utility and accuracy of the ecoregion concept to account for biotic variation, and they cite a handful of other empirical studies that have done so as well (e.g., Whittier et al. 1988; Lyons 1989; Tate and Heiny 1995). In a similar manner, Hawkins and Norris (2000) dedicate an entire journal issue to examining the performance of different landscape classifications for aquatic bioassessments. These studies illustrate the importance of checking the ecological relevance of classifications used for conservation purposes

Closing Remarks

Steps seven and eight are arguably the most important considerations in the creation of ecosystem classifications because they reveal classification errors and uncover how useful the classification will be when it comes to achieving management goals (e.g., identifying and protecting biodiversity). Yet there are very few examples of classification systems that are validated at all (e.g., Feminella 2000 and references therein, Hawkins et al. 2000); moreover, validation by necessity entails testing the classification against the appropriate criteria that relate back to the management goals, and to our knowledge, this is almost never done.

It is not without some chagrin that we have omitted the importance of biotic variables such as community types, species interactions, access, and dispersal, which can influence stream ecology and limit species diversity. It is widely known that abiotic processes are functionally inseparable from biotic ones, especially in stream networks because they are longitudinally connected (Leopold et al. 1964; Poole 2010). The reasons for this omission are twofold: first, biotic data are often unavailable for multiple taxonomic groups in many regions of the world and we did not wish to emphasize any particular taxonomic group. Second, there still seems to be a lack of knowledge on the subject of how biotic and abiotic processes interact (Power et al. 1988; Jackson et al. 2001; Jones 2010).

The proposed eight considerations are the result of our synthesis of the recent literature and the perceived need for greater understanding of directional aquatic networks given regional variability at multiple spatial scales. This approach emphasizes the importance of abiotic process regimes in time and space that form a template for flowing waters. The outline of considerations may serve as a guide for researchers and managers set on the road to creating practical AECs. We contend that widely separated aquatic systems with similar flow, temperature, nutrient, and sediment regimes will be similar ecologically and will have biota with similar functional and life history traits. We also contend that species patterns will be subject to regional variability due to dispersal barriers, post-glacial access routes, differences in the species pool, and taxonomic variability, and there will always be atypical aquatic systems (e.g., those subject to unusual disturbance dynamics) that will defy classification (Olden and Kennard 2010).

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