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Abstract

This report discusses the concept of minimum instream flow (MIF) as a means of regulating average flows and variation in flow levels in rivers, with special reference to the James River basin in Virginia. Different instream-flow methods are reviewed to establish general criteria that may be applicable to the James and other Virginia rivers. Natural-resource economics, Virginia's water policy, and drought-management plans for the James River and other basins in the state are examined for their ability to facilitate MIF implementation. We tentatively suggest that 30-40%, 15-20%, and 10%, respectively, of the mean annual flow (MAF) will be required to implement drought-watch (voluntary conservation), drought-warning (mandatory restriction of nonessential water uses), and emergency (mandatory rationing) regulations for MIF management during summer droughts. We also recommend the adoption of a statewide water management plan that incorporates longer-term drought policies, including changes in the statutory and institutional framework, to proactively manage instream-flow levels and water uses in response to increased population growth and development. Because riparian and estuarine ecosystems may require higher average flows than stream biota need, and because such ecosystems and riparian wetlands require periodic flooding to maintain the productivity and life cycles of their biotic components, we tentatively suggest higher MIFs during the nonsummer seasons. For example, for maximum protection, criteria such as 60-100% of MAF as a nonsummer MIF, 200% of MAF for spring flushing every year, and 300-500% of MAF every 5-10 years for spring flooding might be considered.

However, before any criteria are formally adopted, results from additional instream-flow research, as well as priorities for other instream and offstream demands, must be evaluated. Furthermore, a more holistic stream classification system will be needed in Virginia to refine these flow-percentage criteria (MIFs) and to prioritize water-protection needs.

Keywords: Minimum instream flow, instream-flow methods, Virginia, James River basin, aquatic ecosystem, riparian ecosystem, water law, water policy, drought management.

1 Minimum Instream Flow

1.1 Definition

For this report, we define MIF as the minimum discharge of water needed in a particular season to sustain a given instream use or combination of uses, such as water quality, fish and wildlife populations, recreational activities, navigation routes, and aquatic habitats. River basin and watershed management for developing and maintaining water supplies must integrate offstream uses—public water supply, irrigation, livestock watering, electric power generation, and industrial cooling—with instream uses sustained by these minimum discharges.

Unfortunately, MIF is not a single, unambiguous discharge that managers can use for all rivers in every situation. This is because of the differences among rivers, their legal and institutional settings, and the variety of instream uses possible (Milhous and Grenney, 1981; Gernerd, 1982; Mosley, 1983, 1985; CDM, 1986; Gore, 1989; Jackson et al. 1989; Brown et al., 1991). Stream habitat diversity and site-specific instream and offstream requirements mean that judgment and interdisciplinary and interagency communication, including biologic, hydrologic, geomorphic, and management assessments, are required to effectively evaluate flows and protection alternatives. Even if a manager could focus on only one instream use (e.g., fish-assemblage protection), one MIF would be inadequate because seasonally varying flows are necessary for maintaining habitat availability and providing cues for such activities as migration, spawning, and feeding.

How can a water manager satisfy both instream and offstream users? Can one MIF be defined for a given season that will allow multiple water uses? To answer these important questions, we must address the history of the MIF concept.

1.2 Background and Progress of MIF Research

The MIF concept originated in the United States with instream-flow analyses in the early 1970s to assess damage to instream resources in the West (Orsborn and Allman, 1976a, 1976b; Stalnaker and Arnette, 1976; Tennant, 1976; Wesche and Rechard, 1980). Offstream users often removed all of the water from smaller streams in this region, leaving dry stream beds and causing complete losses of economic, aesthetic, and recreational values (Anderson, 1982; Milhous and Anderson, 1983; MacDonnell et al., 1989; Belsey and Herbst, 1990). Such dewatering and diminished values have been particularly prevalent in the Colorado River basin (Elfring, 1990; Gray, 1991). Fights to preserve instream flows on federal lands also have been intense (Goldfarb, 1985; Tarlock, 1986; Gawthrop, 1987; Stadtmore, 1987; Gelt, 1988; Jackson, 1988; Robinson, 1988; Anonymous, 1989a; Lamb and Lord, 1992).

Instream-flow management programs are spreading in Virginia and other states in the East and West (e.g., CDM, 1986; Estes and Harlan, 1987; Filipek et al., 1987; Gelt, 1988; Jensen, 1988; Reed and Meed, 1988; MacDonnell et al., 1989; Collins, 1990; Burkardt, 1992). Instream-flow assessment techniques will be used increasingly in the eastern United States as water shortages in densely populated, drought-prone areas and rapid fluctuations in stream flow from hydropower generation become more common (Bain and Boltz, 1989). Documentation of the deleterious effects of extended low flows on water quality, hydrologic cycles, riparian animals and plants, aquatic biota, and estuarine organisms is growing (Copeland, 1966; Orsborn and Allman, 1976a, 1976b; Rozengurt and Herz, 1981; Mosley, 1983, 1985; Gilliland et al., 1985; Howarth, 1988; Woessner and Potts, 1989; Stromberg and Patten, 1990; Tyus, 1990).

Both the Tennant method and instream flow incremental methodology (IFIM) have been implemented in state programs, the former for general (reconnaissance), conservative estimates of MIF, and the latter for more detailed, site-specific MIFs. These assessment tools have helped promote compromises between instream and offstream users, particularly because their incremental bases allow various flow scenarios to be evaluated, rather than inflexibly specifying one MIF value (Tennant, 1976; Stalnaker, 1977; Doerksen and Lamb, 1979; CDM, 1986; Cavendish and Duncan, 1986; Garn, 1986; Lamb, 1986; Brown et al., 1991).

Early instream-flow research with the IFIM and other methods focused upon salmonid fish in western U.S. streams (Stalnaker and Arnette, 1976; Tennant, 1976; Wesche and Rechard, 1980; Annear and Conder, 1984; Bovee, 1986; Conder and Annear, 1987). Now, the methods are being applied extensively throughout the United States to studies of warm-water and cool-water stream fish (Wesche and Rechard, 1980; Orth and Maughan 1981b, 1982; Sale et al., 1982; Leonard and Orth, 1988; Bain et al., 1988; Bain and Boltz, 1989; Orth and Leonard, 1990; Aadland et al., 1991), macroinvertebrates (Gore and Judy, 1981; Orth and Maughan, 1983a; Morin et al., 1986; Keup, 1988; Statzner et al., 1988; Bain and Boltz, 1989; Jowett and Richardson, 1990; Jowett et al., 1991), and riparian plants and animals (Gilliland et al., 1985; Latka and Yahnke, 1986; Stromberg and Patten, 1990, 1991).

1.2.1 Office/Discharge Methods

The earliest form of instream-flow protection in the United States was the 7Q10, (i.e., the 7-day low flow expected every 10 years, on average), developed by civil engineers. This flow was designed to protect water quality and not biological organisms. For this reason, other more biological instream-flow methods typically yield higher MIFs (Hayes and Watson, 1984; CDM, 1986; Orth and Leonard, 1990). The 7Q10 also does not differentiate between low flows caused by overdraft of water and those caused by natural droughts. This differentiation could be important because the water quality of drought-caused low flows is generally better—the lack of precipitation and runoff causes reductions in toxicants and nutrients (Muchmore and Dziegielewski, 1983). Despite these limitations, 7Q10 continues to be used by Virginia and several other states (Reiser et al. 1989; Burkhardt, 1992). These criticisms also apply to the 1Q30 (VWCB, 1988), the critical drought-indicator flow, since this flow is even lower than the 7Q10.

The MIF methodology was modified when biological researchers helped develop discharge (office) techniques. In a review of these methods, CDM (1986) concluded that the New England (aquatic-base flow) and Tennant (Montana) methods are the best for estimating MIFs in Virginia, although the latter method is more empirically based and, therefore, more defensible. The New England method estimates the low-flow MIF from the median (50% exceedance) flow during the month of lowest flow (August or September), and uses this throughout the nonspawning period of August-March (Leonard et al., 1986). Flows during the spring spawning season of April-July can be protected by using median historical flow for each month separately as the MIF.

In contrast, the Tennant method is incremental (Tennant, 1976; Orth and Maughan, 1981a). This promotes greater flexibility in specifying the percentage of mean annual flow (MAF) necessary for whitewater and other types of boating, for flushing out fine sediments, and for maintaining biological and other recreational resources in conditions ranging from optimal to severely degraded. Higher flows (at least 30-40% of MAF) usually are specified during the flooding and/or spawning season (e.g., January or March to June) for the James River basin (CDM, 1986; Leonard et al., 1986; Orth and Leonard, 1990) and Oklahoma streams (Orth and Maughan, 1981a, 1982). In contrast, summer-fall flows often are set at 10-20% of MAF, although Tennant (1976) originally specified higher summer-fall than winter-spring flows. These flow criteria were developed based upon qualitative examination of stream habitat at different flows by Tennant (1976) and others (e.g., Orth and Maughan, 1981a), and, thus, are subjective and not necessarily reproduceable.

The Tennant method also has been criticized because it may not apply readily to streams of differing size, shape, hydrology, and geography, particularly since it may overestimate flow needs during summer (CDM, 1986; Virginia Power, 1986). Virginia Power also suggested that average flow be replaced by median flow, because most streams have heavily skewed flow patterns that may cause Tennant's method to overestimate necessary flows. However, this ignores the fact that Tennant (1976) worked with midwestern streams having skewed flow regimes to derive his flow criteria. The Tennant method also appears to overestimate MIF relative to the IFIM for larger streams (Leonard et al., 1986; Orth and Leonard, 1990), and is expected to underestimate MIFs for recreational purposes in smaller streams. This is because all available flow may be required to create and maintain adequate whitewater, velocities, depths, and widths for water sports (e.g., Mosley, 1983, 1985; CDM, 1986). Indeed, Brown et al. (1991) cite a study showing that, for every doubling of MAF across rivers, canoeing zero flow (the flow below which a canoe scrapes bottom excessively) increases by 50%, rather than by the 100% expected for a constant Tennant percentage.

1.2.2 Field/Office Methods

Field/office techniques were developed later to quantify instream-flow needs for fish, recreation, water quality, and other uses through hydrologic and biologic research and computer simulations (Orsborn and Allman, 1976a, 1976b; Stalnaker and Arnette, 1976; Wesche and Rechard, 1980).

The habitat variables measured often are not the same for different field/office methods, and the variables used may not be the same even within methods such as the IFIM (Orsborn and Allman, 1976a, 1976b; Stalnaker and Arnette, 1976; Bovee and Milhous, 1978; Smith, 1979; Wesche and Rechard, 1980; Trihey and Wegner, 1981; Bovee, 1982; Theurer et al., 1982; Scott and Shirvell, 1987). Depth, velocity, substratum size, substratum embeddedness (sedimentation), cover, stream width, wetted perimeter, oxygen, and temperature are most frequently assessed.

CDM (1986) determined that the best of these methods were the wetted-perimeter and IFIM methods, particularly the latter, which has become well-developed. The former is a hydraulic-rating method and typically is applied to riffles, where wetted perimeter usually decreases much faster with lower flows than in pools (e.g., Kraft, 1972; Litton, 1984; Carter et al., 1985). The MIF is the threshold flow, below which wetted perimeter declines rapidly, although the method gives ambiguous values for certain channel shapes, and the inflection point is not necessarily a reasonable MIF (Annear and Conder, 1984; CDM, 1986; Milhous, 1987; EA, 1991). The wetted-perimeter method also assumes that riffle habitats are most important as food-producing areas. This is probably true for small salmonid streams (Waters, 1972; Keup, 1988), but not for larger streams where pool habitats predominate and riffles may be absent. Nevertheless, based on the importance of backwater and side-channel habitats in larger streams for fish and aquatic invertebrates (Amoros, 1991; Schlosser, 1991; Vadas, 1992; Vadas and Orth, in progress) and the disappearance of these habitats at low flows (Carter et al., 1985), the wetted-perimeter method is potentially useful in larger streams when edge habitats are sampled. Brusven and Trihey (1978) found this method useful for examining the effects of hydropower generation on water-level fluctuations and on the stranding of flowing-water (lotic) invertebrates. Annear and Conder (1984) developed a statistical variant of the method for more objectively determining MIFs. Several other hydraulic-rating methods focus on depth and/or velocity to ascertain whether anadromous salmonids and boats can negotiate riffles (Stalnaker and Arnette, 1976; Wesche and Rechard, 1980; Mosley, 1982; CDM, 1986; EA, 1991).

The IFIM is a habitat-rating method because it involves collection of biologic and hydraulic data and extensive habitat simulation (Bovee and Milhous, 1978; Trihey and Wegner, 1981; Bovee, 1982, 1986; Milhous et al., 1989). It is, therefore, more relevant to aquatic ecosystems and organisms than hydraulic-rating methods because it has a biotic, as well as a hydraulic, component, and requires multiple transects to better sample aquatic habitats (Bovee, 1982; CDM, 1986; Loar et al., 1986; Armour and Taylor, 1991). The IFIM also relies on empirically determined flow criteria rather than the more arbitrarily assigned flow criteria of the Tennant method (Mosley, 1983; Loar et al., 1986).

The biotic component consists of habitat suitability index (HSI) models for species and life stages, based upon field data and/or general knowledge. These models are used to estimate optimal microhabitat types for resident fish or aquatic invertebrates.

The hydraulic component is PHABSIM (physical habitat simulation), a set of computer programs that simulate hydraulic conditions and (with recent modifications) water quality at different flows, based upon field data and river hydraulic principles. In tandem, PHABSIM and HSI models are used to estimate the habitat availability for fish at different flows. Such analyses are complicated by the specific, possibly non-overlapping habitat needs of different species and life stages, causing researchers to choose the habitat-use guild, species, or life stage most sensitive to low flows for MIF estimation. Habitat-use guilds have been employed to lump species and/or life stages, and mesohabitat types have served to combine microhabitats. These tools simplify instream-flow analyses and MIF determinations in streams with high habitat and piscine diversity (Morhardt et al., 1983; Bain et al., 1988; Leonard and Orth, 1988; Bain and Boltz, 1989; Aadland et al., 1991; EA, 1991; Vadas, 1992).

The IFIM, however, has been heavily criticized by Mathur et al. (1983, 1985, 1986) and others (Smith, 1979; Annear and Conder, 1984; Conder and Annear, 1987; Bleed, 1987; Irvine et al., 1987; Scott and Shirvell, 1987), and subsequently defended by Orth and Maughan (1983b, 1986). Much of the criticism concerns details (e.g., the hydraulic assumptions of PHABSIM, the use of aggregative HSI indices, and the absence of water quality and other environmental variables from these models) rather than the method's general approach. The criticisms stimulated improvements in IFIM, and the newest IFIM package allows more flexibility in PHABSIM and HSI modeling to correct most of these problems (Herricks and Braga, 1987; Milhous, 1988; Milhous et al., 1989). More recent efforts to simplify PHABSIM and HSI analyses by examining changes in mesohabitat or macrohabitat availability at different flows, rather than more detailed microhabitat analyses, may further improve the method and reduce output and informational overload (Annear and Conder, 1984; Smith and Carswell, 1984; Vadas and Orth, in prep.). However, researchers appear evenly divided—they either consider the IFIM to be too complicated for general use or too simplified for application to larger spatial scales of analysis (Milhous et al., 1987b; Armour and Taylor, 1991).

Another criticism is that the IFIM has been verified only for internal (model) consistency and not validated with long-term data sets of animal abundance in respect to flow changes (Terrell and Nickum, 1984; Armour and Taylor, 1991), but this means that the method is untested rather than invalid (Reiser et al., 1989). Perhaps the more important criticisms are that IFIM analyses (a) do not work well for high-gradient streams (Armour and Taylor, 1991); (b) cannot predict changes in aquatic vegetation, sedimentation, and channel morphology in alluvial, unstable rivers (Bleed, 1987; Milhous, 1988); (c) do not explicitly incorporate biotic variables such as predation and competition (Orth, 1987); and (d) predict changes in habitat availability rather than changes in fish population abundance, because the many factors affecting animal abundance make populations fluctuate stochastically (Milhous and Grenney, 1981; Graholm et al., 1985; Morhardt and Altourney, 1985; Loar et al., 1986; Gore and Nestler, 1988; Tyus, 1992). Nevertheless, streams of various morphologies and sizes and those subjected to different types of environmental impacts are amenable to PHABSIM analyses (Bovee and Milhous, 1978; Bovee, 1982; Stalnaker and Milhous, 1983; Trihey and Baldridge, 1985; Trihey and Stalnaker, 1985), and modeling of fish population dynamics relative to instream flow should improve biological predictions of the IFIM (Williams, 1984; Mattice, 1990).

1.3 Comparison of Instream Flow Methods

The most defensible, reliable, and accurate instream-flow methods appear to be the Tennant, New England, wetted-perimeter, and IFIM (CDM, 1986), but even these four methods often yield different MIFs in at least some months of the year (Jensen 1988). Orth and Maughan (1982) found the best correspondence among the four methods during low-flow months (July-December) when designating fish-assemblage MIFs in an Oklahoma stream. This MIF was 5-10% of the MAF, as opposed to the 30% level that they had recommended based on the Tennant and IFIM methods for a spawning season of April-June (Orth and Maughan, 1981a). Newcombe (1981) used an early version of the IFIM to recommend 60-100% of MAF during the spring spawning and rearing season for salmonids, thus corroborating Tennant's (1976) optimal-flow criteria.

For chinook salmon spawning and hydraulic variables (depth and velocity), Estes and Osborn (1986) obtained crude MIF estimates of 171% and 50% of MAF for the IFIM, with and without substratum size as a variable. Apparently, depth and velocity were inadequate for defining habitat preferences, because an office method using these two variables also underestimated MIF (115% of MAF). Annear and Conder (1984) found that the wetted-perimeter method gave high fish MIF estimates for summer-fall flow in trout streams, relative to Tennant's (1976) 30% "good" criterion. The IFG-4 version of the IFIM overestimated MIF in small streams, but underestimates were typical in larger flowing-water systems. Such underestimates may have been caused by the use of average, instead of demersal (near-bottom), velocity as the variable (Annear and Conder, 1984); somewhat higher MIFs have been estimated when using demersal velocity rather than average velocity (CDM, 1986). This discrepancy may result from fish hovering near the bottom, causing average velocity to be an inappropriate measure of their habitat use.

Further comparisons of methods are needed, and these are being made in southwestern Virginia on the Roanoke River (Vadas and Orth, in progress). Nevertheless, similarity of results does not validate these methods. Validation currently is being addressed through examination of long-term biological changes resulting from flow alterations, because time lags are more likely for larger fish species (Orth and Maughan, 1982; Orth, 1987; Mattice, 1990). Conversely, differences between methods need not imply that one method is better than another because methods were designed to address particular instream-flow needs. For example, the 7Q10, New England, and other (wetted perimeter, IFIM, or Tennant) methods are designed, respectively, to provide good water quality, a semi-natural flow regimen, and aquatic habitat.

The MIF concept is needed to equitably allocate instream flows and maintain beneficial instream uses. Instream-flow methods continue to be refined, and are being incorporated into long-range, water-resource plans in western and eastern states. Clearly, no one best method for instream-flow analysis is currently available, particularly because of the unintentional biases associated with the different methods (Lamb, 1989). The lack of availability of biological and recreational data, rather than the generally accessible hydraulic data, probably limits the accuracy of these analyses (Gore, 1989; Tyus, 1990; Brown et al. 1991). Improvements in instream-flow methodology that make it more defensible, realistic, site-specific, and time-efficient will promote MIF implementation by state and federal organizations (Railsback et al., 1990).

1.4 Problems of MIF Implementation and Socioeconomic Analyses

Traditional offstream water uses must be factored into any MIF decision. Increasingly, MIFs are recognized as necessary for preventing excessive withdrawals from flowing waters, especially during droughts. Negotiated compromises incorporating long-term offstream and instream values and uses need to be made, and these should be important components of water-supply plans or comprehensive state water policies. Balancing instream and offstream uses can best be addressed by optimizing the marginal economic benefits of flow diversions, regulated releases, and instream uses. This approach requires modeling the effect of different river flows on various recreational and offstream uses (Loomis, 1987a).

Instream users also must specify particular MIFs for different amounts of damage to their resources (Brown et al., 1991).

Because the Tennant and IFIM methods can be used as incremental tools (i.e., MIFs can allow a certain degree of instream damage, such as good versus excellent fish habitat availability), they are well-suited for compromise solutions. For example, Orth and colleagues (Leonard et al., 1986; Orth and Leonard, 1990) multiplied their IFIM-based MIF by fractions (20-100% in 20% increments) to provide five flow estimates that could be used by water managers to allocate flows. These researchers and Annear and Conder (1984) also proposed that flows yielding at least 90% of maximum area (habitat availability) for a target species should signify the effective MIF. In contrast, Domingue et al. (1989) relied on exceedance values relative to the optimal habitat availability of 50-90% in 10% increments, and EA (1991) used both weighted-usable-area and habitat-exceedance increments. CDM (1986) suggested that higher instream flows could be implemented in streams with valuable fishery, aesthetic, or other recreational resources, when states such as Virginia expand their lotic classification systems beyond trout streams and public water supplies.

Such expanded classification in Virginia and other states is occurring because of the Environmental Protection Agency's (EPA's) requirement under the Clean Water Act section 303 (C) (4) and the agency's water quality standards regulation (40 CFR 131.32) to include a three-tiered approach to maintaining and protecting various levels of water quality, as a component of each state's water quality standards. States must establish a Tier III category for surface waters of exceptional recreational or ecological community significance and those with exceptional environmental settings. As of July 1, 1993, 23 streams and rivers have been nominated for consideration as exceptional waters in Virginia. However, if incremental-flow criteria are part of the selection and management process, no information is available to determine which flow criteria are most appropriate.

The problem of how to allocate water fairly and maintain reasonable MIFs requires resolution. In addition to competing offstream and instream uses, water resource managers must consider tradeoffs between competing instream uses. Several researchers (e.g., Gillilard et al., 1985; Lampe and Colston, 1986; Duffield et al., 1992) incorporated the economic values of instream flows into more traditional cost-benefit analyses for water development. Assigning a dollar value to various flow activities is the most common and most subjective approach to quantifying instream benefits. Unfortunately, the most common method is not necessarily the best method, and benefits may be more realistically expressed as available area of river bed for fish habitat, or as boater-days per season at different flows. However, these units are not comparable, and only by using similar units can expected benefits be compared under alternative river or stream-regulation schemes and subsequently protected with MIFs (Sale et al., 1982).

Socioeconomic analyses are needed to obtain an optimal balance. Contingent-valuation (CVM), travel-cost (TCM), hedonic pricing, hedonic-travel-cost (HTC), household-production, local-economy, and other methods now are being applied to determine the worth of stream fisheries and other aquatic and riparian recreational resources because of the threats of siltation and deforestation (Theurer et al., 1985; Braden et al., 1989), hydropower development (Sale et al., 1982; Olson et al., 1985; Bishop et al., 1990), and water withdrawals (Gilliland et al., 1985; Hansen, 1986; Loomis, 1987a; Loomis and Cooper, 1990; Brown et al., 1991; Crandall et al., 1992; Loomis and Creel, 1992). Economic evaluations also have been used to assess the worth and feasibility of irrigation systems on rivers (Gilliland et al., 1985; Taylor et al., 1985; Steenhuis and Allee, 1989; Bosch and Broomhall, 1990; Duffield et al., 1992), interjurisdictional water transfers (Cox and Shabman, 1984), water removed for electric power generation (Gilliland et al., 1985) and run-of-the-river lakes and reservoirs (Loomis 1987b, 1989; Khatri-Chhetri and Hite, 1990; Corps, 1992). Compensation of anglers for suboptimal flows by increased stocking of fish (Loomis and Cooper, 1990) or by construction of better passage facilities at dams for anadromous fish (CDM, 1986) can be addressed with economic analyses, as can protection of threatened and endangered species (Boyle and Bishop, 1987). Nevertheless, anglers may not necessarily prefer to be compensated with stocked fish if they desire natural fishing experiences or want the fish population's natural genetic structure maintained. Several of these individual research efforts address only one aspect (e.g., recreation or irrigation) and, thus, are not sufficiently holistic to allocate water.

Brown et al. (1991) reviewed the literature on the relationship between stream flow and recreation, and their interpretation of 25 instream-flow studies (table 1) suggests that recreational quality increases with flow up to a certain level, then decreases with additional flow. Unfortunately, most economists have not quantitatively specified the flow required by recreationists, or have not related their results to mean annual or median flows that are used for office instream-flow methods. Nevertheless, Amirfathi et al. (1985) estimated that 30-50% of the mean flow for a high-water year was required by anglers in some Utah streams. Values below 30% were detrimental to recreational uses, and those above

40-50% significantly diminished the economic returns (low marginal benefit) from recreationists. Moreover, flows below the 25% level were five times more valuable to anglers than farmers (Loomis, 1987a). Walsh et al. (1980) showed that anglers, rafters, and kayakers in Colorado, respectively, required 35%, 40-45%, and 50% of maximum bank-full flows for optimal recreation, including adequate water in the rapids for whitewater boating and enhanced wetted perimeters to avoid human congestion. Flows above the 65% level yielded no marginal benefits. Brown and Daniel's (1991) analysis indicates that 44-59% of average bank-full flows would optimize shoreline recreational opportunities in a regulated Colorado stream. These flows contrast with Daubert and Young's (1981) lower estimates in the same stream for shoreline recreation (18-30%) and fishing (8-18%), but not for whitewater boating (above 44%).

Because these various percentages were standardized to flows higher than the MAF, the results suggest that Tennant's (1976) 30% (good) flow will not adequately protect many instream recreational uses. Indeed, the study by Duffield et al. (1992) of two Montana streams produced optimal recreational flows of 121-161% MAF, and diminished benefits were noted above 191-209% MAF. Based on Flug and Montgomery's (1988) recreational analyses and flow data for the New River at Hinton, West Virginia (Ward et al., 1990), 38%, 25%, and 42% of MAF would be required for optimal whitewater rafting, recreational boating, and fish habitat, respectively.

Other economic analyses have focused more on agriculture than water withdrawals. Gilliland et al. (1985) estimated that 89% of yearly flow (for MIFs of 75% MAF in January-April, 116% in May and November, and 27% in summer-fall) would minimize habitat losses for channel catfish (14% loss) and for whooping cranes during spring (8% loss) and fall (no loss). In contrast, irrigation-caused lower flows (61% of yearly flow) would result in habitat losses of 62% for catfish, and of 17% (spring) and 75% (fall) for whooping cranes, with only a 3% increase in agricultural yield. On the other hand, Bosch and Broomhall (1990) estimated diminished losses in annual net returns for irrigation of 1%, 16%, 46%, and 64%, respectively, if summer MIF was 4% (7Q10), 10%, 20%, and 30% of MAF in the Pamunkey River of eastern Virginia. Perhaps summer MIFs above Tennant's 10% (poor) level harm farmers enough to mandate economic analyses of instream-flow benefits, although a few studies have documented the greater value of instream uses relative to agriculture and other offstream diversions at low flows (Loomis, 1987a; Colby, 1990).

In total, natural resources researchers have used seven economic valuation techniques: CVM, TCM, hedonic pricing, HTC, household-production, gross-expenditure (GEM), and market-price-appraisal (MPAM) (Sorg and Loomis, 1985; Hansen, 1986; Ward and Loomis, 1986; Anonymous, 1987; Decker and Goff, 1987; Kaiser et al., 1987; Swanson et al., 1989; Walsh et al., 1989). CVM and TCM have been especially popular with federal agencies, because these techniques are older, better tested, and not overly data-intensive. These methods are best examined with the willingness-to-pay (WTP) approach, because of the problems with the counterpart approach, willingness-to-sell, for determining natural-resource values (Gregory, 1986). Both CVM and TCM estimate net WTP (i.e., the consumer surplus, the maximum cost that recreationists would be willing to pay over and above the actual cost they already pay to use a given natural resource) (Sorg and Loomis, 1985). Environmental managers perhaps should focus on net WTP for predicting potential monetary losses that would occur if products of the natural resource cannot be developed, because actual money currently paid typically is transferred into other channels (i.e., saved or spent elsewhere) and is not considered an economic loss (Sorg and Loomis, 1985). In contrast, methods that do not estimate net WTP (e.g., GEM), or those that are less broadly relevant to public resources (e.g., MPAM), are less reliable for valuation of fish and wildlife resources (Sorg and Loomis, 1985; Kaiser et al., 1987; Swanson et al., 1989; Walsh et al., 1989).

CVM and TCM are similar in their reliance on in-person or telephone interviews and mail surveys. CVM is easier to employ, but TCM is used more often and is better accepted by academia because it is based upon real, rather than hypothetical, money (Kaiser et al., 1987; Loomis, 1987a). CVM slightly underestimates net WTP because only primary (recreational) activities are considered; whereas TCM also incorporates secondary (other) activities during recreational trips (Walsh et al., 1989). The major methodological difference, however, is that TCM and other market-based methods (e.g., hedonic pricing) are based upon the actual costs to recreationists of using natural resources, whereas CVM is a simulated-market method that is based upon monetary bids provided by recreationists. TCM estimates net WTP indirectly from knowledge of actual costs and total (gross) WTP, whereas CVM estimates net WTP directly by the bidding process.

CVM, nevertheless, has an important advantage for assessing the value of natural resources. Namely, market-based methods cannot estimate the option, existence, and bequest values of water and other natural resources (e.g., the value of future trips to the river, of knowing that the river has adequate water to be healthy and visually pleasing, and of protecting the river for future generations, respectively) (Loomis, 1987a; Sanders et al., 1990). TCM and hedonic-pricing methods may underestimate WTP as much as five times below the actual value of the resource in instream-flow analyses. Therefore, CVM is usually more appropriate and more commonly used to study resources such as instream flows (Loomis, 1987a, 1987b; Colby, 1990). The two methods produce similar monetary results, however, when benefits to only recreation (not preservation) are factored into instream-flow and other economic analyses (Sanders et al., 1991). Clearly, there are tradeoffs between the two valuation techniques.

TCM estimates net WTP by examining travel and entry costs to and from recreational sites (price = P) under the reasonable premise that such costs reflect the recreational value of these trips. Based upon visitation rates of people to the site (quantity = Q), a demand curve (P versus Q) can be generated, from which consumer surplus is estimated indirectly. This method was improved by incorporation of proxy (adjustment) factors, to better predict visitation rates (Sorg and Loomis, 1985; Ward and Loomis, 1986; Swanson et al., 1989; Walsh et al., 1989). Consideration of in-state versus out-of-state travel, travel time, roundoff errors when travel distance is estimated in aggregate from the center of a zone (for zonal TCM), demographic and socioeconomic variation among the recreationists, site quality, site substitutability, and desirable human density at sites can improve TCM estimates of net WTP. For example, improving a recreational site could cause congestion, and TCM would overestimate net WTP because it ignores such density dependence. However, no solution is available for the problem of allocating actual costs when a recreational trip is made for more than one purpose (e.g., for fishing at a given site and for visiting relatives), such that the actual cost associated with natural resource use is less than the cost of the trip. The two types of TCM—zonal and individual observation—have advantages and disadvantages, and the latter is applied when recreationists make multiple visits to sites (Ward and Loomis, 1986).

CVM estimates net WTP via one of three major bidding processes, in which recreationists are asked the amount they would be willing to pay for access to a given site. These include single-bid, iterative-bid, and dichotomous-bid formats, the latter requiring that individual recreationists be given individual bids that they can accept or reject as their response (Walsh et al., 1989). Usually, individual components of a given trip are enumerated in combination (as with TCM), because recreationists typically are unable to decompose their trips into individual parts (e.g., the number of fish caught versus the quality of boating).

The three major bidding formats differ in their biases, such that researchers must carefully word their interviews or surveys to avoid four major biases. Strategic bias results when respondents give false information for deceptive purposes. The other three biases are artifacts of presentation resulting from the initial information or bid given by the researcher, from survey and interview instruments (particularly when the respondent gets confused or pressured by the bidding game), and from payment vehicle (some cause more emotional responses than others). All of these biases can affect the respondent's final bid and net WTP estimate. CVM's ability to estimate net WTP thus can be increased by considering proxy factors for payment vehicle, bidding format, and several of the proxy factors used in TCM analyses (Walsh et al., 1989). For example, CVM may underestimate net WTP if single-bid or iterative-bid formats are used, thus requiring larger proxy factors than for dichotomous-bid formats (Walsh et al., 1989). Likewise, entrance fees and sales taxes may cause underestimates and overestimates of net WTP, respectively, because sales taxes spread the costs over greater numbers of people.

Both CVM and TCM are further complicated by different motivations of different recreationists, even within a single user group, such as anglers or canoeists. For example, different people may be fishing for different fish species, and individuals may prefer different human densities on the river (e.g., Knopf et al., 1973; Popadic et al., 1984). Input from economists and social scientists is necessary for such survey and interview techniques to be properly designed and useful for determining the consumer surplus and cost-effective MIFs that satisfy instream and offstream users of various kinds.

Since politicians and water managers often think in socioeconomic terms, biologists and recreationists should be motivated to estimate values of their instream needs with benefit-detriment analyses (Decker and Goff, 1987; Loomis, 1987a). Such analyses may reduce conflicts and polarization between different water-user groups (Colby, 1990). Unfortunately, misconceptions about the true value of water and, thus, conflicts between users, will continue unless accurate data on water use and supply become available (Cox and Shabman, 1984). Economic analyses may facilitate and encourage private protection of instream flows through market transactions, and may supplement or replace the public-protection mechanisms currently favored (Huffman 1986). Natural-resource economics is still a young discipline, and its evolution and refinement should promote more objective and consistent MIFs.

2 Water Resources Management in Virginia

2.1 Virginia's Water Policy

Water allocation is affected by a state's water policy (or lack thereof), and, as clarification, we will review the legal basis for MIF implementation. Virginia and most eastern states rely on the riparian doctrine (common law) and permit systems for allocating water; most western states are subject to the appropriation doctrine (Doerksen, 1977; Butler, 1985; Dixon and Cox, 1985; CDM, 1986; Lamb and Lovrich, 1987; Meyer, 1987; Kundell, 1988; MacDonnell et al., 1989; Cox, 1991). California and a few other states have incorporated both of these doctrines and permit systems into their water policies (Gawthrop, 1987; Stadtmore, 1987; Dziegielewski et al., 1991). The riparian and appropriation doctrines, respectively, give riparian landowners exclusive rights and senior users (first in time) superior rights to remove water from streams for "reasonable and beneficial" uses.

While neither doctrine addresses MIFs specifically, legislative modifications and judicial decisions have strengthened their use as MIF protectors, including condemnation of riparian areas for public purchase and/or development of permit systems in the East, private transfers of water rights in the West, the public-trust doctrine and special water designations (wild, scenic, recreational, or exceptional) in the East and the West, and several other federal and state environmental laws in both regions (Lamb and Doerksen, 1978; Gernerd, 1982; Hayes and Watson, 1984; Lamb, 1984; Bagley et al., 1985; Dixon and Cox, 1985; Trelease, 1985; ASAE, 1986; Ausness, 1986; CDM, 1986; Johnston, 1986; Meyer, 1987; Milhous et al., 1987a; Gelt, 1988; Jensen, 1988; Jackson et al., 1989; MacDonnell et al., 1989; Belsey and Herbst, 1990; Born et al., 1990; Stern, 1990; Cox, 1991; IWR, 1991a). Several western states also have modified their appropriation doctrines to conserve water and limit supplies to junior users during droughts (Walker et al., 1989). Unfortunately, exemptions of eastern farmers from reporting their irrigation uses, and grandfathering senior users' water rights in both the East and West, have compromised the ability of state agencies to effectively manage instream flows. Nevertheless, Virginia farmers reported irrigation use on a voluntary basis from 1982 through 1990, and, in 1991, such reporting became mandatory.

In Virginia, the common law or judge-made law of surface water and groundwater rights was inherited from England and supplemented by Virginia court decisions. Unless expressly abolished or annulled by the Virginia General Assembly, common-law rights remain in effect (Va. Code

§ 1-10). Virginia has applied the common law of riparian right, a type of real-property right, to allow reasonable use of surface waters. This right is contingent upon the ownership of the land through which or by which the waterway flows. The rights in Virginia to use percolating groundwater (most of Virginia's groundwater moves through the soil and the subsurface to underlying aquifers) belongs to the owner of the land surface. The owner can make any use of this groundwater, and has no liability for any harm that use may cause nearby landowners.

Although interpretations of the riparian doctrine formerly relied on natural-flow theory to maintain natural water conditions, increased public demand on supplies has caused adoption of the reasonable-use theory (Dixon and Cox, 1985; Kundell, 1988). Under reasonable use, each riparian owner has an equal right to use the surface water as long as its quality or quantity is not harmfully reduced by this use. A riparian owner can obtain judicial relief for violation of riparian rights only by demonstrating substantial real or threatened damage. Riparian rights are uncertain because they are not lost by simple nonuse, and usually cannot be fixed in magnitude. The individual right may change with new uses or changes of existing uses by other riparian owners. Because riparian rights are considered real property, they may be bought and sold with, or separately from, the land with which they are associated. Before 1989, few statutes had been enacted by the General Assembly to abrogate these rights. The early exceptions were the State Water Control Law, which provided express limitations on the amount and characteristics of wastes that riparian owners may discharge to the state's waters (Va. Code § 62.1-44.16-44.19), and the Groundwater Act, which imposed a system of permits and certificates on common-law groundwater rights to protect limited groundwater supplies in designated groundwater management areas (Va. Code § 62.1-44.93). Under the Water Control Law and the Groundwater Act, common-law rights became subject to state control. According to Burkhardt (1992), the Groundwater Act, if properly implemented, may indirectly protect instream flows in designated management areas.

In 1989, the General Assembly enacted two laws that signaled a slight erosion of the supremacy of common-law riparian rights in Virginia (Collins, 1990). These laws, the Virginia Water Protection Act and the Surface Water Management Areas Act, established the Virginia Water Protection Permit (VWPP) and the Surface Water Withdrawal Permit, respectively. Although the legislation establishing the VWPP program and directing the Virginia Department of Environmental Quality (VDEQ), formerly the Virginia Water Control Board, to develop the regulations was passed in 1989, the regulations were not formally adopted until March

1992 because of numerous public comments concerning the proposed regulations on nontidal wetlands and instream flows (VWRRC, 1992).

A VWPP is required for any project that needs a § 401 certification (Va. Code § 62.1-44.15.1). Section 401 certification is named for a provision in the federal Clean Water Act (FWPCA § 401; 33 U.S.C. § 1341). Under this legislation, before obtaining a federal license or permit, an applicant engaging in any activity that may produce discharges into navigable waters must obtain a certification that this discharge will not impair water quality. The VWPP includes the § 401 certification, and any monitoring requirement or stipulations for effluent or other conditions for compliance with the State Water Control Law and the Clean Water Act usually are part of the federal license or permit. A VWPP is required for any dredging, filling, or discharge of any pollutant into or adjacent to surface waters, and alteration of the physical, chemical, or biological properties of surface waters.

The adopted regulations, which went into effect in May 1992, require local government certification approving any regulated project before VDEQ can issue a VWPP. VDEQ first must issue its VWPP before any federal agency, including the U.S. Army Corps of Engineers and the Federal Energy Regulatory Commission, can issue their permits or licenses. However, certain withdrawals have been grandfathered: no VWPP is required for water withdrawals occurring before July 1, 1989, withdrawals receiving § 401 certification before January 1, 1989, and withdrawals receiving a § 401 certification after July 1, 1989, but before the effective date of the regulations (provided a MIF consistent with the act was included).

The Virginia Water Protection Permit Act applies statewide, whereas the Surface Water Management Areas Act applies only to designated areas in the state where the demand for surface water exceeds threshold limits (Va. Code § 62.1-242 et seq.). The enabling legislation gave the VDEQ authority to declare surface water management areas and to issue permits to regulate withdrawals in those areas during periods of low flow. VDEQ may designate a surface water management area (Va. Code § 62.1-246) where: (1) a stream has substantial instream values as indicated by evidence of fishery, recreation, habitat, cultural, or aesthetic properties; (2) historical records or current conditions indicate that a low-flow condition could occur that would threaten important instream uses; (3) current or potential offstream uses contribute to or are likely to exacerbate natural low-flow conditions to the detriment of instream values; and (4) the public welfare, health, and safety require that regulatory efforts be initiated. In acting on a permit application, the VDEQ is required to balance offstream and instream uses, and, where needed, impose conditions to protect instream uses from unacceptable adverse effects.

In designated surface water management areas, withdrawals of at least 300,000 gallons per month from streams, or bodies of water fed by streams, are regulated by either certificates or permits. About 900 users of 300,000 gallons or more of surface water per month already report their withdrawals to the VDEQ, and it is expected that the number of users affected by the management area permitting program will be less than this number. Users who began withdrawals before July 1989 must apply for withdrawal certificates, which will include water conservation plans. All withdrawals initiated after July 1, 1989, must be permitted. The regulations will rank withdrawals by a priority system based on beneficial use, and low-priority uses will be the first curtailed during droughts.

Permits must include a MIF value for the protection of instream beneficial uses. Both instream and offstream beneficial uses were defined in 1989 by the General Assembly (Va. Code § 62.1-10(b)). Instream beneficial uses include, but are not limited to, the protection of fish and wildlife habitat, maintenance of waste assimilation, recreation, navigation, and cultural and aesthetic values. Offstream beneficial uses include, but are not limited to, domestic (including public water supplies), agricultural, electric power generation, commercial, and industrial uses. Public water supply for human consumption is considered the highest priority.

When flows decrease below this MIF, withdrawals must be stopped or reduced in accordance with permit requirements. Conditions that may be included in the permits are maximum withdrawals, times of day or year during which withdrawals may occur, and requirements for voluntary and mandatory conservation measures. Before issuing, denying, or modifying an application, VDEQ must consider (Va. Code § 62.1-248(B)): (1) the number, object, extent, and necessity of withdrawals on a stream; (2) the nature and size of the stream; (3) the relationship of the activity to the users; (4) the relationship of the necessity of the use and the extent of any detriment caused; (5) the effects on beneficial uses; and (6) any other relevant factors.

The 1992 assembly passed legislation (HB 201) requiring applicants for surface water permits to obtain certificates from local governments showing that projects will comply with land-use ordinances. No management areas could be declared until 1993, six months after the regulations go into effect. The VDEQ is accepting nominations for management area designation, and has received requests for the Richmond and Maury River (Rockbridge County) areas of the James River basin, part of the upper Rappahannock River watershed, the North River near Harrisonburg, the South Fork of the Shenandoah River in Page County, and the Shenandoah River in Clarke and Warren counties.

Virginia's water policy has not been comprehensive and satisfactory to all parties, as evidenced by several controversial public hearings and committee reports to address MIFs, water transfers, and water-supply needs in the last two decades (Cox and Shabman, 1984; Butler, 1985; Kull et al., 1985; Moreau, 1987; Roth et al., 1988; MPI, 1992). Those organizations directly involved in water-quantity and drought issues include VDEQ, four other state agencies (departments of Health, Agricultural and Consumer Services; Emergency Services; and Mines, Minerals, and Energy), Office of the State Climatologist, Corps, U.S. Geological Survey, and National Weather Service. According to Cox and Shabman (1984) and IWR (1991b), the transfer of water-planning authority from the Virginia Department of Conservation and Economic Development to VDEQ in 1972 shifted the focus from water quantity to water quality, forcing local governments in eastern Virginia and the Corps to manage water supplies piecemeal.

A diversity of agencies, committees, and individuals have participated in resolving these issues during the last decade. They include the York River Basin Committee, Roanoke River Water Flow Committee, Friends of the Roanoke River, Appomattox River Water Authority, Governor's Commission on Virginia's Future, Virginia Council on the Environment, Virginia Department of Conservation and Historic Resources, Virginia Drought Monitoring Task Force, State Water Plan Advisory Committee, State Water Commission, Virginia Water Resources Research Center, Virginia Department of Game and Inland Fisheries, Virginia Tech Chapter of the American Fisheries Society, Virginia Farm Bureau Federation, Virginia Agribusiness Council, Virginia Corporation Commission, Virginia Manufacturers Association, Virginia Association of Realtors, Virginia Power, Virginia's Office of the Soil Conservation Service, League of Women Voters in Virginia, the state of North Carolina, Chesapeake Bay Foundation, Environmental Defense Fund, and many Virginia citizens, municipalities, and counties. Clearly, special and diverse interests are concerned with instream flows and other water issues in Virginia.

Although interbasin transfers occur in several states, including Virginia, Kundell (1988) found that no eastern state issued withdrawal permits specifically for water transfers to another river basin. And although several transfers now occur in Virginia, these have had little legislative or judicial support (Cox and Shabman, 1984). Specifically, there have been 12 interbasin and 8 interdrainage transfers of surface water in Virginia since 1988, including 7 interbasin transfers involving the James River and nearby basins (York, Roanoke, and Chowan) and two interdrainage transfers between the lower James River and the tributary Appomattox River. Several other interbasin transfers between the James River and nearby basins (also including the Rappahannock) have been proposed, particularly the highly controversial transfer from Lake Gaston (Roanoke River basin) to the lower James River for use by Virginia Beach and southeastern Virginia (Cox and Shabman, 1984; Kundell, 1988; Waite, 1990; Burkhardt, 1992; MPI, 1992). Interdrainage transfers in the James River basin also have been proposed to supply other areas of the coastal plain, but environmental concerns, water-use conflicts, inadequate water quality in the lower James River (from pollution or salinity), and questionable population-growth estimates have so far prevented any transfers (MPI, 1992).

Without a statutory basis, however, such water transfers need not be environmentally or economically justifiable to affected riparian landowners, and need not allow landowners to receive adequate compensation for lost water-supply opportunities (Cox and Shabman, 1984). Cox and Shabman (1984) emphasize that such a statutory basis should increase the quality, but not necessarily the number, of water transfers in Virginia. This seems a reasonable conclusion, as long as the legislation does not force water-rich localities to give up their water, even if compensation is guaranteed. In any case, the evolution from natural-flow to reasonable-use theory in the riparian doctrine has made water transfers more likely in eastern states (Kundell, 1988), although water transfers for municipal use in Virginia more strictly adhere to natural-flow theory (Burkhardt, 1992).

The 1992 General Assembly considered a bill that would have allocated 60 million gallons of Roanoke River water per day for public water supplies in southeastern Virginia communities (VWRRC, 1992). This bill was carried over to the 1993 General Assembly, but was not passed (VWRRC, 1993). Such near-acceptance of water transfers by the General Assembly further signals the need for an integrated, statewide plan and policy on growth, development, and use of natural resources. Otherwise, conflicts will erupt between rapidly developing, populated, water-deficient areas and more rural, unpopulated, water-rich areas. The Virginia Commission on Population Growth and Development has begun to address certain of these complex issues. Short-term water supply benefits to water-poor areas must be weighed against long-term socioeconomic, aquatic and riparian habitat, water-quality, and ecological changes (Meador, 1992) in water-rich areas of Virginia. As Kundell (1988) noted, future transfers will become controversial because water-rich areas are becoming rare.

In summary, Virginia's water policy is becoming more comprehensive as the riparian doctrine is modified to allow substate permitting systems and water transfers from rivers and other surface waters (Collins, 1990; Waite, 1990; Burkhardt, 1992). Virginia must have the authority to control both existing and new uses of water, the divisions of water among present and future users, and the capability to allocate resources to new uses as the needs change. Business is generally not in full favor of riparian rights because of the uncertainty about water supplies; long-range certainty is critical to business enterprises. The new permitting systems should promote the welfare of individual water users, allow cities and towns to plan effectively for the future, and enhance the development and expansion of industry and commerce.

The Virginia Water Protection Permit and the Surface Water Withdrawal Permit are postulated to represent an eroding of the supremacy of common-law riparian rights in Virginia. Eastern states, including Virginia, may be moving toward applying a legal framework akin to the public-trust doctrine, where all waters of the state are state property, held in trust for all citizens, who have a common right to use the resource. However, the General Assembly is cautious in its actions affecting riparian rights. For example, one section (Va. Code § 62.1-253) of the act establishing Surface Water Withdrawal Permits clearly states,

> *Nothing in this chapter shall be construed as altering, or authorizing any alteration of, any existing riparian rights except as set forth in permits issued pursuant to this chapter. The conditions in such permits shall be in force only in those times when low stream flows, or the potential therefore, results in a declaration as provided for in subsection A of § 62.1-249.*

Although change is occurring, instream-flow and riparian-wetland protection and the use of the public-trust doctrine for recreational preservation still are controversial issues in Virginia and other states. It is, therefore, still difficult to implement effective MIF strategies, allocate water fairly, and integrate the management of surface water and groundwater and of water quality and quantity (Cox, 1968; Hayes and Watson, 1984; VWCB, 1988; Waite, 1990; IWR, 1991a, 1991b; Burkhardt, 1992).

2.2 Virginia's Drought Policy

MIF protection is especially crucial during droughts; therefore, a state's drought plan is clearly relevant to instream-flow protection. At the national level, drought protection has become a priority because socioeconomic and environmental well-being are greatly affected by low flows and by the unavailability of resources due to pollution (Dziegelewski et al., 1991; IWR, 1991a). Water-use efficiency and effective conservation strategies (via mandatory regulations and/or a reward system) are needed to protect instream-flow resources (Grima, 1979; Lee, 1981; USDA, 1988; Walker et al., 1989). Voluntary conservation alone is not the answer because it will not reduce water use by more than 20-25% (CDWR, 1988). However, no consensus exists on whether mandatory conservation regulations or water-rate increases and drought surcharges are the most effective means of reducing water use below that of voluntary conservation. According to Burkhardt (1992), Virginia Beach's various water conservation measures have curtailed per-capita water use by only 25%.

Agricultural and irrigation systems are particularly vulnerable to drought (Vellidis et al., 1985; Anonymous, 1989b; Dziegelewski et al., 1991), but their water demands can be substantially reduced through water conservation practices (Rait and Lieuwen, 1989). In Virginia, drought-management plans have been prepared for the Potomac River basin (IWR, 1991a; Steiner, 1991; Burkhardt, 1992) and are underway for a portion of the James River basin (IWR, 1991b). The state as a whole, however, is still inadequately prepared for future droughts (Walker et al., 1989; IWR, 1991b), although improvements in monitoring techniques are being made (Loganathan et al., 1985; Vellidis et al. 1985). For example, three critical drought flows (1Q30) have occurred in the James River basin since 1898—in 1930, 1932, and 1966 (VWCB, 1988; IWR, 1991b). Several municipalities in the upper, middle, and lower James River basin predicted water-supply deficits by 1990, 1988-2030, and 1988-2000, respectively, in their late-1980s analyses (VWCB, 1988; Waite, 1990).

Drought plans should be tiered (CDM, 1986), with certain instream-flow levels to indicate drought watches (voluntary conservation), warnings (mandatory restriction of nonessential outdoor water uses such as landscaping and car-washing), and emergencies (mandatory rationing, with drinking water and other essential uses taking precedence over other offstream and instream uses). Hence, a comprehensive system of priorities of water-use categories must be in place before droughts occur, so that each user knows the order in which to apply water restrictions.

Unfortunately, Virginia and many other riparian doctrine states have not adopted priority systems.

State, county, and local agencies in Virginia, particularly in the eastern part of the state, have short-term (tactical/contingency) drought plans, but most states (including Virginia) do not have effective long-range (strategic) plans (Moreau and Little, 1989; Dziegelewski et al., 1991; IWR, 1991a, 1991b). Tactical plans, formulated by about 25 states and half of the utilities in the southeastern United States, use the existing laws and institutional framework and include emergency conservation and reallocation, drought relief funds, and cloud seeding. In contrast, strategic plans require changes in the laws and framework, including the building of new reservoirs for water storage and aqueducts for water transfer, new building-code regulations to require water-conservation devices for plumbing fixtures, improved irrigation systems (e.g., drip replaces spray) to reduce waste, water-recycling requirements for car-wash businesses, and zoning laws to prevent local overdraft of water supplies. Clearly, tactical and strategic plans must replace crisis management if droughts are to be handled effectively (Wilhite and Wood, 1985).

All of the long-range solutions have been implemented in some areas of Virginia, although statutory basis for protection of future reservoir sites via condemnation and compensation (eminent domain), desalination of brackish water, water reuse in agricultural and industrial facilities, and management of human populations and immigration are lacking (VWCB, 1988; IWR, 1991b). The coastal plain, including the lower James River area, has been particularly affected by droughts because of salt-water intrusion into groundwater wells and relatively rapid population growth and development (VWCB, 1988; USSCS, 1990; Waite, 1990; IWR, 1991b; MPI, 1992). Unfortunately, officials in these counties and municipalities, as well as the Corps and VDEQ, have dismissed the importance of tactical and strategic water conservation in their estimates of future water needs for southeastern Virginia (Waite, 1990). Their preference for designing water supplies to avoid conservation during droughts is not a realistic long-term solution (Waite, 1990), particularly because drought-management plans must satisfy all user groups and not just municipalities (Moreau and Little, 1989). In summary, many short- and long-range drought-management plans facilitate MIF protection, but ultimately, planning for and managing population growth and development will be most important, particularly in the eastern third of the state and in the Roanoke-Salem urban area (Roth et al., 1988; VWCB, 1988; MPI, 1992).

The ultimate consequences of drought are measured by its effect on social and economic activities (Dziegelewski et al, 1991; IWR, 1991a). As human populations and their activities continue to increase, and water demands continue to grow, even minor droughts can have serious impacts. State plans for population growth and economic development must incorporate a flexible and functioning water-management plan applicable to droughts. Because droughts affect larger geographic areas than a single community, the ability of any one community to act is affected by the actions of other communities in the drought area. Therefore, planning must be done on a regional or statewide basis. In the past, Virginia farmers have been opposed to large-scale drought planning because of their strict adherence to the riparian doctrine (Walker et al., 1989). Obviously, the cooperation of this influential group of water users is crucial to successful statewide drought-management efforts.

3 The James River Basin

3.1 Resources

The James River basin drains 25% (26,400 km^2) of the land area of Virginia, has a MAF of 300-350 m^3/s at its mouth, and traverses over 550 km from its headwaters in the Allegheny mountains of eastern West Virginia to the Chesapeake Bay at Hampton Roads (Woolcott, 1985; Abbott et al., 1988; VWCB, 1988; EA, 1991; IWR, 1991b; Corps, 1992; Garman and Nielsen, 1992). The James comprises the largest basin and has more tributaries than any other Virginia river, its waters encompassing 700 km^2 of surface area (figure 1). In the Piedmont and Coastal Plain provinces (areas with the most critical drought problems), the middle and lower James are alkaline, warm (1-33 °C), and of moderately low gradient (0.5 m/km on average) (EA, 1991; Garman and Nielsen, 1992). The MAF at Richmond is 211 m^3/s (EA, 1991).

The Upper James is located in the Ridge and Valley and Blue Ridge provinces. This watershed is more forested than the middle and lower James, with warm-water conditions (maximum 31 °C) in the main stem and cool-water conditions (below 28 °C) in the tributaries (Leonard et al., 1986; Leonard and Orth, 1988). Rocky substrata (gravel to bedrock) predominate, the gradient is moderate (1.2-2.3 m/km), and the average main-stem discharge is 45 m^3/s below Lick Run. The stream bottom is less embedded with fines (mud and sand) than downstream, as reflected by higher crayfish densities in the upper James (Mitchell and Smock, 1991).

Although waste-treatment facilities have improved water quality since the 1970s, dioxin contaminates the upper James River (above Lynchburg), several contaminants (fecal coliforms, heavy metals, and pesticides) occur in the middle James, and salinity is too high for potability in the lower James (below Hopewell, 100 km from the Chesapeake Bay), according to Garman and Nielsen (1992) and MPI (1992). Kepone levels have declined in the water column, and water quality has improved sufficiently for the 13-year fishing ban to be lifted from the lower James River and its estuary (Woolcott, 1985; Garman and Nielsen, 1992). Thermal pollution from 7 active power plants has not significantly affected fish abundance and composition, but the 12 hydropower dams have limited migrations of anadromous fish (Woolcott, 1985; EA, 1991; Garman and Nielsen, 1992; VDGIF et al., 1992). No major impoundments are located on the James River, but 29 water-supply basins, 3 major urban areas, 15-20 industrial/municipal facilities, and extensive farming occur along the river (VWCB, 1988; IWR, 1991b; Garman and Nielsen, 1992).

The basin's fishery is economically valuable for both sport fish—black bass and other centrarchids—and commercial fish, especially clupeids (EA, 1991; Garman and Nielsen, 1992). A total of 101 fish species representing 20 families—including 3 endemic species and subspecies, introduced bait and sport fish, and several estuarine forms—have been collected in the basin (Raney, 1950; Hocutt et al., 1986; Leonard et al., 1986; Garman and Nielsen, 1992). Anadromous fish, including clupeids and percichthyids, are important, and upstream and downstream migrations of adults and downstream movements of juveniles occur during most of the year (March-January) (CDM, 1986). Cyprinid fish are also important, and apparently are exclusive hosts for an endangered mussel in the upper James watershed (Hove and Neves, 1989). These fish are dominant in abundance and species richness except below the fall line, where centrarchids and other game fish are more important (Leonard and Orth, 1988; EA, 1991; Garman and Nielsen, 1992). The smallmouth bass is the important sport fish in the upper and middle segments of the river, whereas the largemouth bass replaces it in the tidal area (Randolph, 1992). The mean fish biomass of 41 kg/ha in the middle and lower sections is below that of other centrarchid/ catostomid-dominated river basins on the southeastern Atlantic slope (Garman and Nielsen, 1992), but total fish density is, nonetheless, high (EA, 1991).

The James River basin has other important natural, recreational, and cultural resources (Heerwald, 1983; Woolcott, 1985; CDM, 1986; Abbott et al., 1988; EA, 1991; IWR, 1991b; Corps, 1992). Several sections of the river are protected as historic/scenic areas, byways, parks, and wildlife preserves and refuges at the state and national level. The middle James River offers excellent boating opportunities, and the scenic section of the river above Richmond has large whitewater falls that attract expert and less advanced boaters, depending upon the flow level. Swimming, picnicking, hiking, horseback riding, hunting, and nature study are also important recreational activities in the James River basin. Forested lands predominate, followed by agricultural lands, making open space more important than urbanized areas. Other important activities in the lower James River watershed include mining for clay, sand, and gravel; public and private oyster harvesting (the best in Virginia); other shellfish harvesting; and commercial shipping. Riparian wetlands are abundant; hunting of large and small game, including waterfowl, is productive; and several threatened/endangered/special-concern species of plants and animals—aquatic, semiaquatic, and terrestrial—occur in the basin. In the lower James River, bald eagle habitat may be the best in the continental United States.

3.2 Instream-Flow Research

Orth and colleagues (Leonard et al., 1986; Orth and Leonard, 1990) used the 7Q10, Tennant, New England, wetted-perimeter, and/or IFIM methods to estimate MIFs in four streams of the upper James River watershed above Lynchburg (figure 1). MIF estimates were dissimilar for the five methods (table 2) because of differing methodological assumptions. The New England method overestimated early-spawning flows relative to the IFIM because high flows are assumed necessary for successful reproduction. The New England MIFs for nonspawning seasons, the single estimates for the 7Q10 and wetted-perimeter methods, and the Tennant estimate for autumn were lower than those for the IFIM. The Tennant method and IFIM were most in agreement for the early-spawning season at all sites except the upper James River. As with other streams in the Valley and Ridge province of Virginia (CDM, 1986), the 7Q10 yielded flows near Tennant's poor level, namely, 10% of MAF.

Because the streams yield different seasonal MIF patterns with the IFIM approach, for this publication we took an overall average across months for each stream to yield a single summer MIF (table 2). We also calculated low and high MIFs for each stream based on the five fish habitat-use guilds defined by Leonard and Orth (1988): IA (slow riffle and run), IB (fast riffle), II (pool and run), III (pool), and IV (shallow pool). The guilds yielding the median and highest MIFs were used to specify the fall (low) and early spring (high) flow needs, respectively. The MIF for each guild was calculated by the average optimal wetted usable area (WUA) for each life stage and species of fish studied (Leonard et al., 1986, table 14). The type II and IB guilds typically defined the low- and high-flow MIFs, respectively.

Based on these results, the smallest stream needed the largest percentage of MAF, whereas the largest stream needed the least, to satisfy low- and high-flow requirements. Clearly, the Tennant method's year-round recommendations of 30% and 100% of the average flow to maintain good versus optimal stream conditions compares favorably with the IFIM results for the fall versus early spring MIFs, respectively. However, research is necessary to verify that somewhat smaller or larger percentages are needed for MIF implementation in streams of different sizes, particularly, before recommending lower MIFs in larger streams. For example, Orth and Maughan (1982) found the summer MIF for the IFIM to be only 10% of MAF for a midsized stream (MAF = 10.5 m^3/s) in

Oklahoma, and Annear and Conder (1984) concluded that the IFIM underestimates MIF in larger streams. Interestingly, the median IFIM estimates of MIF for the three larger sites of Leonard et al. (1986) were 59-74% of mean summer flows (July-September). These estimates resemble those of Narayanan (1986), whose summer criteria to optimize benefits for shoreline fishing and other recreational activities were 61-84% of summer flow in a Utah river. In contrast, Dunlap Creek and the middle James River at Richmond yielded relatively high (177%) and low values (38%), respectively.

Using the same upper James River data set and a hybrid approach of the IFIM and New England methods, Orth (1990) made MIF recommendations for the Maury River near Buena Vista (table 2). This method for MIF estimation is probably more realistic and practical than either of its component methods because it allows maintenance of semi-natural, seasonal fluctuations in flow (unlike the IFIM) and lower MIFs in the spring and summer than the overestimates obtained by the New England and IFIM methods, respectively. For most spring months (March-May), Orth (1990) prescribed 90% exceedance flows for each month. These MIFs were higher than optimal IFIM flows, and, thus, valuable for spring flushing functions, but not as high as the likely overestimated New England spring MIFs. In contrast, Orth (1990) recommended median (50% exceedance) flows for each month separately during summer (July-November). These flows were lower than optimal IFIM flows and somewhat higher than the New England MIF, but, nevertheless, more typical of the river at these times. Finally, for most winter months (December-February) and June, he recommended the optimal MIF, based upon IFIM analyses.

The Jackson River, a tributary of the upper James River, has been examined with other MIF techniques below Lake Moomaw, near Warm Springs. This reservoir (Gathright) has been required to release at least 35-43% MAF (4.5-5.5 m^3/s) in October-April, 52-54% MAF (6.6-6.9 m^3/s) in May and September, and 60-63% MAF (7.6-8.0 m^3/s) in the summer, based upon data presented by EA (1991) and Prugh et al. (1990). These MIFs were set to protect water quality and public water supplies for Covington, located downstream of the reservoir. These flow criteria are in the good to optimum range defined by Tennant (1976), but unusual in specifying higher flow needs during the summer than the colder seasons.

Several instream-flow methods also have been used in the middle James River watershed. Ivy Creek, a tributary near Lynchburg (figure 2) has been subjected to the Tennant method using the 30% criterion (CDM, 1986). We followed Orth (Leonard et al., 1986; Orth and Leonard, 1990) in using 40% and 20% of MAF for the high-water (midwinter to early summer) and low-water (midsummer to early winter) seasons, respectively. Just below Ivy Lake reservoir, the respective MIFs are thus 0.11 and 0.06 m^3/s, whereas those farther downstream (where the stream crosses Route 621) are 0.22 and 0.11 m^3/s.

Various MIF methods have been applied to the James River at Richmond, as shown in table 3. The study section includes both a main and side channel (figure 3). The latter (Kanawha Canal) is an artificial waterway with an MAF of 20 m^3/s (Prugh et al., 1990); however, with the cessation of hydropower generation, recent canal flows have averaged only 10 m^3/s to protect channel stability and potential recreational uses (CDM, 1986; EA, 1991). The MIFs in table 3 reflect the flows needed at the downstream end of Richmond, near the 12th Street and Hollywood hydroplants (figure 3). As done with the upper James data and for the purposes of this publication, we estimated two fish-assemblage MIFs, based on seven fish habitat-use guilds derived from EA's (1991) data: shallow-slow, shallow medium-fast (madtom and chub), medium-shallow fast (jumprock), deep-slow, deep-fast, moderate-flow anadromous (herring), and high-flow anadromous (shad). The 20% and 40% MAF Tennant criteria were taken from Leonard et al. (1986).

The Richmond single-value MIFs are disparate, reflecting the differing needs of instream-user groups and the different instream-flow techniques (table 3). The lowest value is the 7Q10, as expected, whereas aesthetic requirements are somewhat higher, but still below 20% of MAF. Boat-passage, fish-passage, and fish-assemblage requirements were higher, but less than 40% of MAF. Water-contact and shoreline activities, recreational boating, and multiple-use MIFs were moderately high, but below 65% of MAF. The highest MIFs were for whitewater boating, as expected, and these required 40-135% of MAF. These MIFs are consistently lower than those of Garn (1986) for a western U.S. river, where brown trout (47-60% of the MAF, calculated by IFIM), acoustic aesthetics (67% of MAF), and water quality (7Q10, 51% of MAF) required at least half of the average flow. These results suggest that the aesthetic MIF for the James River did not account for whitewater sound production. The western river required more water for waste assimilation because of its higher summer flows, and more of its flow was needed to support fish, perhaps because it was a small stream (MAF = 2.1 m^3/s). Interestingly, the fish-assemblage MIF for Richmond corroborates arguments that larger streams (at least in the James River basin) need relatively less flow (near 20-30% of MAF) to support fish (CDM, 1986; Leonard et al., 1986; Orth and Leonard, 1990).

The Richmond seasonal MIF methods were also discrepant (table 3). Much like the results for the upper James River, the Tennant and New England methods were most dissimilar during the early-spawning season, because of the high flows specified by the latter method. The New England method, however, yielded low-flow estimates near the 20% level of the Tennant method, similar to the upper James River results (tables 2 and 3) and consistent with CDM's (1986) findings for various Virginia streams (aquatic base flow averaged 21% statewide and 22% in the James River basin). The two techniques usually gave much lower flows than the U.S. Fish and Wildlife Service (USFWS) estimates, which were based upon median monthly flows for December-April, multiple-use needs (including anadromous-fish passage) during May-September, and multiple-use needs other than fish passage during October-November. The USFWS method produced unrealistic values, because MIF was never below 65% of MAF, even though lower summer flows occur naturally. Indeed, fish migrations may not require flows of 40% MAF, and vernal flows probably can be lowered without major harm to stream ecosystems (Bowles, 1983; CDM, 1986; Orth, 1990; EA, 1992).

On ungaged streams in New England and North Carolina, low-flow and high-flow MIFs have been estimated as 0.3-0.6 and 2-4 cubic feet per second per square mile (cfsm) of drainage area, respectively, but such criteria vary with physiographic province within these regions (Morhardt and Altourney, 1985; Moreau and Little, 1987; Kulik, 1990). Somewhat lower values were encountered at the five James River sites in tables 2-3. These sites yielded aquatic base flows of 0.13-0.23 cfsm, increasing in value from the smaller to the larger sites (Dunlap Creek to the middle James River). Likewise, using the fish guild with the highest MIF at each James River site to estimate high-flow needs, we obtained actual values of 0.4-1.6 cfsm, with the smallest values for the two main-stem sites. Because of this intrabasin variability in flow criteria, we recommend that ungaged streams in the James River basin not be subjected to the New England method.

3.3 Flow Criteria for the James River and Other Virginia Waters

3.3.1 Summer/Fall Flow Needs

Until further instream-flow results for fisheries, wildlife, recreation, and other instream uses are attainable (e.g., the ichthyofaunal study of the upper Roanoke River, Vadas and Orth, in progress), we tentatively suggest that maintenance of a summer MIF of 30-40% average discharge should serve as a drought-watch flow, below which voluntary conservation should be implemented (CDM, 1986). Perhaps flows below 15-20%

of MAF should be warnings for mandatory restrictions of nonessential water uses, and flows below the 10% level should be emergency flows requiring mandatory rationing. Based on these projections, the VDEQ's Ware Creek flow criteria of 6.5% and 3.25% for voluntary and mandatory conservation, respectively, do not adequately protect aquatic resources (Waite, 1990). Similarly, Moreau and Little (1989) suggest that Georgia's use of 1.2 and 0.5 times the 7Q10 as flow criteria for, respectively, restricting and banning nonessential outside water uses, is much too low.

The criteria that we tentatively propose for the James River probably would lead to voluntary water conservation every summer, partial mandatory restrictions during some summers, and mandatory rationing during occasional summers. The recently completed upper Roanoke River study (Vadas and Orth, in progress), is expected to test the usefulness of these drought-management criteria. Several other sources also support the reasonableness of our flow criteria. As examples, two western states—New Mexico and Montana—allow MIFs of up to 50% of MAF if such flows are needed to preserve stream biota (MacDonnell et al., 1989), and MIFs to maintain recreational fishing in southeastern U.S. tailwater streams were 15-55% MAF (Wood and Whelan, 1962). Quantitative aids to drought management need to be developed, implemented, refined, and reimplemented (Moreau and Little, 1989), via monitoring of offstream withdrawals, return flows, and instream flows, as well as precipitation, contribution of runoff, evaporation, groundwater, reservoir storage, and Palmer (drought) indices.

The use of incremental percentage criteria is preferable to water-quantity management via piecemeal regulation of individual users, because these criteria provide management flexibility, may be applied across streams, and can evaluate and predict cumulative drought impacts. If selected flow levels could be maintained over the spawning and recruitment seasons of selected fish species, flow stochasticity would be reduced, with potential benefits to anadromous and resident fish (Schlosser and Karr, 1981; Milhous, 1986; Harvey, 1987; Rulifson and Manooch, 1990; Zincone and Rulifson, 1991). That is, percentage criteria should be used to manage both average flow levels and to minimize detrimental flow fluctuations that are often caused by hydropower generation (Cushman, 1985; Bain et al., 1988; Tyus and Winter, 1992). For example, Wood and Whelan (1962) estimated that the highest aquatic productivity and fishing success would occur in southeastern U.S. rivers with maximum flows maintained below 75-118% MAF. Nevertheless, specifications of MIFs should not be construed as advocating constant flow levels, because seasonal and yearly changes in flows are probably necessary for maintaining community biodiversity, according to the immediate disturbance hypothesis (Stanford and Covich, 1988; Poff and Ward, 1989; Yount and Niemi, 1990). Clearly, additional research is needed to define optimal ranges of flows to protect aquatic ecosystems.

3.3.2 Winter/Spring Flow Needs

Although winter/spring flow needs of fish can be estimated from several office/discharge methods, as well as from the MIF for the high-flow fish guild, other aquatic organisms and watershed resources need consideration to validate these MIFs. In particular, flushing flows in early winter and late spring are important, but not yet accurately quantified. Such flows are necessary to whitewater boaters (Tennant, 1976; CDM, 1986); to the biota of estuarine ecosystems, such as the Chesapeake Bay, for maintaining life cycles of plankton, shellfish, and finfish (Copeland, 1966; Rozengurt and Herz, 1981; Tyus, 1990); to anadromous fish as migration and spawning cues (Huntsman, 1945; Hayes, 1953; Shira, 1976; Rozengurt and Herz, 1981); to aquatic and semi-aquatic animals in riparian wetlands for maintaining their life cycles (Clark, 1979; Crance and Ischinger, 1989); to stream biota for preventing low-oxygen stress caused by the buildup and breakdown of leaves and algae (MacKenthun et al., 1945; Blum, 1956; Larimore et al., 1959; Slack and Feltz, 1968); to habitat integrity by scouring (deepening) pools and preventing sedimentation (Tennant, 1976; Nelson et al., 1987); and to the growth and seedling success (recruitment) of riparian trees (Reily and Johnson, 1982; Fenner et al., 1985; Harris et al., 1987; Stromberg et al., 1991).

Information on flow needs for riparian vegetation and associated wildlife is scant (Lamb and Lord, 1992). Stromberg et al. (1991) estimated that Arizona floodplain trees require 3 times the base flow for vernal germination and 2,300 times the base flow (7-year flood) for recruitment. However, such recruitment flows, if they occurred annually, would damage and kill trees through scouring (Reily and Johnson, 1982; Harris et al., 1987). Stromberg et al.'s (1991) flow criteria are equal to or greater than base-flow ratios for the median fish taxon for Leonard et al.'s (1986) and EA's (1991) fish guilds in James River sites (1.2-3.9), Domingue et al.'s (1989) six fish guilds in Minnesota rivers (1.1-3.4), and Orth and Maughan's (1982) ratio for eight fish species/life stages in an Oklahoma stream (1.3).

Such flood flows also can be compared to MAFs. In the upper and middle James River watershed, flows that are 2,300 times the base flow would equal 300-500% of MAF (tables 2 and 3). Such a flow would represent only 118% MAF in Orth and Maughan's (1982) Oklahoma stream, a value similar to their median monthly flows (115-117%) during February and March and their recommended flushing flow (114%). Decamps et al. (1988) noted that annual flooding for about 30 days occurred in the active (proximate) floodplain of a European river at flows of 300-400% relative to MAF. Every 2-20 years, floods of 700-1,400% of MAF reached the floodplain of importance of this river, and flows above the 2,200-2,300% level reached the exceptional floodplain 3 or 4 times per century as catastrophic floods. Williams (1978) calculated that bank-full discharges flooded the active floodplain every 0.9 years (329 days) on the average, based on data from 51 U.S. streams. Apparently, floods that are 300-500% of MAF occur regularly, and such flows may be important to the life cycles of biotic components of aquatic and riparian ecosystems.

Riparian vegetation requires adequate instream-flow levels throughout the year to prevent stunting, death, reproductive failure, and changes in the species composition of trees and herbs (Harris et al., 1987; Kondolf et al., 1987; Stromberg and Patten, 1990; Smith et al., 1991). Unlike aquatic plants and animals, trees may not need continuous flow, but their average requirements may exceed those of lotic organisms (Stromberg and Patten, 1990). For example, dominant tree species required 80-100% and 40-60% of MAF, respectively, for good growth in Rush (Stromberg and Patten, 1990) and Bishop creeks in California (Stromberg and Patten, 1991).

The well-documented importance of riparian vegetation in removing nutrients and toxicants from runoff, maintaining water quality, moderating water temperatures, and providing food resources and habitats for stream invertebrates and fish (Lynch et al., 1977; Likens, 1985; Beschta and Plotts, 1986; Harmon et al., 1986; Bisson et al., 1987; Woessner and Potts, 1989; Welsch, 1991) and terrestrial wildlife (Verner et al., 1986; Lamb and Lord, 1992) strongly suggests that instream-flow analyses should incorporate the requirements of riparian vegetation. To function effectively, riparian zones (buffer strips) must be 15-30 m in width (Chauvet and Decamps, 1989; Welsch, 1991). Such buffers can retain 80-90% of the sediment and nutrients moving through this vegetated zone and may be required in designated areas of the Chesapeake Bay superbasin (Peterjohn and Correll, 1984; Welsch, 1991). Clearly, floods that inundate the active floodplain and benefit the riparian zone should be a component of state-adopted stream-flow regimens (Decamps et al., 1988; Chauvet and Decamps, 1989; Amoros, 1991).

Similarly, riparian wetlands require adequate flows and periodic flooding to retain and degrade incoming pollutants that would otherwise enter the adjacent stream; maintain high productivity and diversity; and provide food, habitat, and spawning and nursery areas for animals (Clark, 1979; Crance and Ischinger, 1989; Paulson and Davis, 1990). Such wetlands may reduce flood peaks by nearly 100%, thereby decreasing downstream damage to buildings and property as well as natural habitats (Paulson and Davis, 1990). Wetlands function much like riparian forests in reducing the movement of sediment and nutrients into streams (estimated 80-95% retention). Van Haveren (1986) emphasized the importance of detention and retention structures such as wetlands for modulating stream flows (e.g., to prevent extreme floods and droughts). We could find no quantitative studies on the instream-flow requirements (MIFs) of floodplain wetlands, but such data are needed to protect and manage lotic-riparian systems in Virginia and elsewhere.

MIFs to protect the habitat needs of local aquatic biota also do not necessarily maintain downstream aquatic and recreational resources. Stromberg and Patten (1990) estimated that about 100% MAF was needed in their study stream in California to protect the downstream Mono Lake ecosystem. This value was similar to that found for the riparian forest. Loomis and Creel (1992) concluded that flow in the Stanislaus River was nearly an order of magnitude more valuable for protecting recreational resources in the downstream San Joaquin River than when just local recreational needs were considered. Downstream estuaries may require 70-75% of MAF to prevent ecosystem destruction (Rozengurt and Herz, 1981; Davoren and Ayres, 1984).

Possible negative impacts of low flow in estuarine ecosystems may pertain to the fisheries and wetland resources in the James River estuary and Chesapeake Bay (Copeland, 1966; Rozengurt and Herz, 1981; Tyus, 1990), and include: (a) increased barriers to fish and invertebrate migrations; (b) reduced commercial oyster production because of the inappropriate salinities, competition with noncommercial oyster species, and parasitism; (c) reduced production of other commercially important shellfish such as peneid shrimp and blue crab; (d) reduced production of phytoplankton and its consumers, such as zooplankton and fish; (e) destruction of coastal wetlands; (f) reproductive failure of fish and invertebrates; and (g) drastic changes in the species composition of the estuarine community. Such negative effects result from increased salinity, altered hydrological conditions, and reduced influx of riverine nutrients, and can be controlled only by the maintenance of normal spring and fall flooding.

These riparian and downstream flow needs are within Tennant's (1976) optimal range, and contradict the conclusion of CDM (1986) that larger streams can sustain greater flow reductions per capita. Clearly, future researchers must consider larger spatial and temporal scales to properly evaluate the cumulative and indirect impacts of flow reductions on natural resources (Cada and Hunsaker, 1990; Burns, 1991). A MIF to protect aquatic ecosystems during summer is inadequate for the whole year, and, possibly, for riparian and downstream ecosystems and recreational resources even during the summer. We agree with Schlosser (1982), Beschta and Platts (1986), and Phinney and Powers (1991) that habitat protection in smaller streams is necessary to protect larger streams, estuaries, and marine habitats downstream in the basin. Apparently, MIFs that reduce average yearly flows below 70-75% of MAF in tributaries can damage estuaries unless such flow reductions are restricted to certain strategically located, key streams.

The importance of riparian forests, wetlands, and estuarine ecosystems in the James River basin as natural, recreational, and commercial resources supports our proposition that future instream-flow studies should incorporate the water needs of these components as well as aquatic life and wildlife, such as the bald eagle, that depend on this biota (Gelt, 1988). This riparian and estuarine data suggest that nonsummer MIFs should be much higher than summer MIFs, at least during spring and fall floods. We tentatively suggest that, for maximum protection of these habitats, such floods should exceed 200% MAF every year and 300-500% every 5-10 years in the spring season, and nonsummer MIF should be near 60-100% MAF. Such values should be tested and examined in the context of an integrated, holistic framework of long-term offstream and instream uses of the James River.

4 Caveat

This bulletin represents one of a number of pieces of instream-flow information currently being published and disseminated for Virginia. Although a more complete picture is emerging for the waters of the Commonwealth, we recognize that more studies are needed. For example, although our literature and data review is extensive, we have not reviewed *all* relevant data and literature. In particular, we recognize a need to (1) review more papers documenting the flow needs of aquatic, riparian, and estuarine resources; (2) review and compare an expanded set of instream-flow methods; (3) modify these methods to establish three flow criteria per year, namely high-flow (spring), low-flow (summer), and medium-flow (other seasons); and (4) consider maximum, as well as minimum, flow needs to protect fish assemblages, other aquatic life, and the many instream flow uses.

These modifications should provide more robust and realistic bases for setting MIFs, to better protect watersheds without setting unnaturally and unnecessarily high flow levels. Such adjustments and fine tuning should benefit watershed residents and Commonwealth citizens who rely on a diversity of instream and offstream uses.

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Glossary

active (proximate) floodplain - the zone of river bank subjected to annual flooding. Mesic (riparian and wetland) plant species typically occur in this zone, but are not abundant farther from the river.

average velocity - the mean velocity in a water column, typically located 60% from the surface (and 40% from the bottom) in freshwater streams.

demersal velocity - the velocity more relevant to stream organisms that tend to stay near bottom (e.g., macroinvertebrates and fish). This habitat variable is typically taken 1-20 cm above the bottom, depending upon the organism studied, and is sometimes called *nose* velocity.

exceptional floodplain - the upland zone far enough away from the river bank to be inundated only during major spates (e.g., by 25- and 50-year floods).

field/office techniques - instream-flow techniques that require collection of habitat and sometimes biological data, as well as analysis (in the office), to determine MIFs.

50% exceedance - the median flow that is exceeded 50% of the time and, thus, not exceeded 50% of the time. Exceedance flows typically are calculated on a monthly or annual basis.

floodplain of importance - the zone near the river bank that is subjected to periodic flooding, but not every year. The vegetation is composed of a mixture of riparian and upland species.

habitat-rating - field/office techniques that require habitat and biological sampling, often in several representative transects. The relation between the habitat variable(s) and organismal abundance is used to evaluate habitat quality at different flows.

highest guild - the fish guild exhibiting the highest MIF in IFIM analysis of a given stream. Presumably, such MIFs are useful for setting flow levels in higher-flow seasons (i.e., spring and/or fall).

hybrid - the use of more than one instream-flow method to generate MIFs for different months or seasons (e.g., Orth (1990) used a combination of the IFIM and New England method in the upper James River).

hydraulic-rating method - field/office techniques that often require habitat sampling in only one (often considered critical) transect. The habitat variable(s) serves as a surrogate for biological variables, and is used to evaluate habitat quality at different flows.

incremental - instream-flow techniques that can be used to reach compromises among river-user groups (e.g., the IFIM and Tennant methods). Such methods can be used to specify several flow levels that provide different degrees of protection for instream resources.

individual-observation TCM - this travel-cost method depends upon unaggregated data for travel distance (i.e., the travel distance for each individual or each travel group is used in the economic analysis).

macrohabitats - larger habitat units in a given stream that encompass several mesohabitats (i.e., at least one riffle-pool sequence). The representative- and critical-reach approaches to IFIM analyses, as well as watershed analyses, are all macrohabitat approaches.

median guild - the fish guild exhibiting the intermediate (median) MIF in IFIM analysis of a given stream. Presumably, such MIFs are useful for setting flow levels in low-flow months (i.e., in the summer).

mesohabitats - moderate-sized habitat units that are reasonably homogeneous in hydraulic and channel-roughness characteristics to be considered valid units for defining fish habitat-use guilds (e.g., fast-riffle, deep-run, and backwater-pool).

microhabitats - small habitat units that may be present in mosaic fashion in a given stream meander (i.e., in contrast to mesohabitat units, a given microhabitat type is more homogeneous in hydraulic and channel-roughness characteristics, and should occur more than once in a given meander).

natural-flow theory - an early version of the riparian doctrine that specified that water withdrawals were illegal if they significantly affected natural flow levels in a given stream. In contrast, the later version of reasonable use allowed withdrawals to affect flow levels, as long as other riparian owners were not significantly harmed by such withdrawals.

office/discharge technique - instream-flow techniques that specify MIFs via analysis of historical hydrological data for a given stream section. Such data typically consist of mean, median, or exceedance flows by month or year, and are often available from the U.S. Geological Survey's gaging stations.

Palmer index (PI) - this index of soil moisture is widely accepted as a drought index, although it only measures water supply and does not take human demands upon water resources into account.

7Q10 - the lowest flow for seven consecutive days that has a 10% chance of occurring in a given year (i.e., the 7Q10 is expected once every 10 years, on average). This flow often is considered the MIF to protect water quality in streams receiving no pollution.

validation - MIFs generated by instream-flow methods such as the IFIM can be tested for accuracy only with independent, experimental data (i.e., an instream-flow method is validated if flow levels below the predicted MIF cause adverse biological effects that the researcher has predicted).

verification - an instream-flow model such as the IFIM can be tested for internal consistency (robustness) using the same data set (e.g., MIFs generated by the full data set can be compared to MIFs formulated from subsets of the habitat data, or fish abundance predicted by an HSI model can be regressed against the data used to build the HSI model to test the goodness of fit).

Virginia Water Control Board (State Water Control Board) - As of July 1993, this state agency became the Water Division of the Virginia Department of Environmental Quality.

zonal-observation TCM - this travel-cost method depends upon aggregated data for travel distance (i.e., all recreationists from a specified zone are assumed to have driven the same distance, to simplify the economic analysis).

Tables

Table 1. Relationships between stream flow and recreation (Brown et al., 1991).

Method	Study: Author (date)	River (state)	Activity*	Dependent Variable
		DIRECT MEASURES OF SITE-SPECIFIC EFFECTS		
Expert Judgment	Van Haveren et al. (1987)	Beaver Creek (AK)	Canoeing	Floatability
	Williams (1991)	Poudre (CO)	Several	Rec. Quality
Systematic Asessment of Flows, Small Sample				
Flows Depicted by Photos	EA Engineering (1990)	Clavey (CA)	Swimming	Suitability
	Litton (1984)	Toulumne (CA)	Viewing	Visual Quality
Controlled Flows Experienced Onsite	Bayha and Koski (1974)	Snake (ID)	Several	Several
	EA Engineering (1991)	McKenzie (OR)	Boating	Suitability
	Giffin and Parkin (1991)	Kennebec (ME)	Rafting, Fishing	Rec. Quality, Safety
User Surveys				
One Experienced Flow Per Respondent	Bishop et al. (1987)	Colorado (AZ)	Rafting, Fishing	WTP/Trip
	Duffield et al. (in press)	Big Hole and Bitteroot (MT)	Fishing, Shoreline	WTP/Day
	Moore et. al. (1990)	Aravaipa (AZ)	Hike, Swim	Preferred Flow
	Shelby et al. (1990)	Gulkana (AKO	Boating	Rec. Quality
Flows Depicted by Photos	Brown and Daniel (1991)	Poudre (CO)	Viewing	Scenic Beauty
	Daubert and Young (1981)	Poudre (CO)	Fish, Boat, Shore	WTP/Day
	Ward (1987)	Chama (NM)	Fishing, Boating	WTP/Season
Flows Described Verbally	Bishop et al. (1987)	Colorado (AZ)	Rafting, Fishing	WTP/Trip
	Narayanan (1986)	Blacksmith Fork (UT)	Camp, Hike, Fish	Visits/Year
	Shelby et al. (1992b)	Colorado (AZ)	Rafting	Quality, Safety
	Vandas et al. (1990)	Dolores (CO)	Boating	Rec. Quality
	Walsh et al. (1980)	Several (CO)	Fishing, Boating	WTP/Mile/Day
Flow Impacts Described Verbally	Harpman (1990)	Taylor (CO)	Fishing	WTP/Year
	Johnson and Adams (1988)	John Day (OR)	Fishing	WTP/Year
Observation of Use at Various Flows	Several	Several	Several	Visitors/Day
Mechanical Measurement	Garn (1986)	Red (NM)	Listening	Decibels
	Hawkins (1975)	Several (UT)	Listening	Decibels
		MODELS		
Conceptual	Milhous (1990)	Salmon (NY)	Several	Suitability
	Nestler et al (1986)	Chattahoochee (GA)	Several	Rec. Quality
Empirical	Corbett (1990)	45 Rivers	Canoeing	Minimum Flow

*Shoreline indicates picnicking, camping, hiking, and relaxing.

Table 2. Optimal MIF recommendations for four study sites (see figure 1) in the upper James River watershed, based upon data presented by Leonard et al. (1986) and Orth (1990, unpub. data). Both discharge (Q, m^3/s) and percentage of average flow are given for the eight instreamflow methods used. The winter (W) season was December-March, early spawning (ES) was April-May, late spawning (LS) was June-July, and fall (F) was August-November.

Method	Season	Dunlap Creek		Craig Creek		Maury River		Upper James River	
		Q	%	Q	%	Q	%	Q	%
IFIM, graphical method	W	2.3	48	2.8	26	5.9	32	10.8	24
	ES	1.7	37	4.0	36	5.9	32	9.1	20
	LS	1.1	24	4.5	42	5.9	32	9.1	20
	F	2.3	48	2.8	26	6.8	36	9.1	20
Seasonal Mean		2.0	42	3.3	30	6.2	34	9.6	21
IFIM, median guild	F	2.3	49	3.2	29	5.5	30	10.5	23
IFIM, highest guild	ES	7.0	148	14.5	133	25.0	135	35.9	79
Tennant	W	0.9-1.9	20- 40	2.2- 4.4	20- 40	3.7- 7.4	20- 40	9.1-18.2	20- 40
	ES	1.9	40	4.4	40	7.4	40	18.2	40
	LS	0.9-1.9	20- 40	2.2- 4.4	20- 40	3.7- 7.4	20- 40	9.1-18.2	20- 40
	F	0.9	20	2.2	20	3.7	20	9.1	20
New England	W	0.6	13	1.8	16	3.7	20	9.0	20
	ES	3.5-4.9	75-104	9.3-12.9	86-119	16.5-22.7	89-122	38.4-54.9	85-121
	LS	0.9-2.2	19- 46	2.4- 5.7	22- 52	5.5-11.4	29- 62	12.6-26.7	28- 59
	F	0.6	13	1.8	16	3.7	20	9.0	20
Wetted-perimeter, all seasons		0.8	18	1.7	16	2.8	15	3.4	7
7Q10, all seasons		0.3	6	0.9	8	1.7	9	5.1	11
Hybrid[a] approach	W	--	--	--	--	5.9- 8.2	32- 44	--	--
	ES	--	--	--	--	7.5-11.6	40- 62	--	--
	LS	--	--	--	--	5.5- 5.9	29- 32	--	--
	F	--	--	--	--	3.7- 5.3	20- 28	--	--

[a]Orth (1990)

Table 3. Optimal MIF recommendations for the middle James River watershed near Richmond (see figure 3), based upon data presented by Bowles (1983), CDM (1986), Clarkson (1992), and EA (1992). USFWS = U.S. Fish and Wildlife, * indicates MIFs generated from IFIM studies, and # indicates MIFs formulated from formal recreational surveys. See table 2 for other abbreviations.

		Tenant		New England		Hybrid (USFWS)	
		Q	%	Q	%	Q	%
Seasonal	W	42-84	20-40	44	21	152-318	72-151
	ES	84	40	196-253	93-120	197-253	93-120
	LS	42-84	20-40	74-118	35-56	197	93
	F	42	20	44	21	140-197	66-93

	7Q10		Recreational boating		Aes-thetics		Boat passage		Multiple use	
	Q	%	Q	%	Q	%	Q	%	Q	%
Single-value methods	19	9	28-61	13-29	23	11	--		85-113	40-54
	--		51-99*	24-47*	--		42-71*	20-34*	--	
	--		59-130#	28-62#	34#	16#	45#	21#	--	

	Anadromous passage		Whitewater boating		Water-contact and shoreline activities		Fish assem-blage, median guild (F)	
	Q	%	Q	%	Q	%	Q	%
Single-value methods	28-57	13-27	99-127	47-60	--		--	
	34-82*	16-39*	159-283*	75-134*	28-85*	13-40*	57*	27*
	--		93-215#	44-102#	48-116#	23-55#	--	

	Fish assemblage, higest guild (ES)	
	Q	%
Single-value methods	--	
	213*	101*
	--	

Figures

Figure 1. Study sites (represented as letters A-D) on four streams in the upper James River watershed (western Virginia) used to generate the MIF recommendations of Table 2. This map is modified from Leonard and Orth (1988).

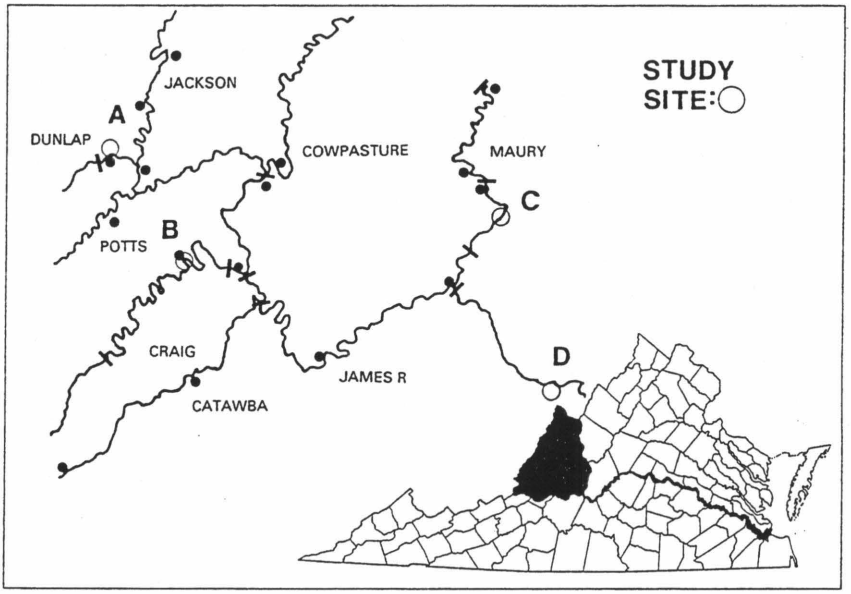


Figure 2. Two study sites (represented as letters A-B) on a tributary (Ivy Creek) near Lynchburg, in the middle James River watershed (central Virginia). This map is modified from CDM (1986).

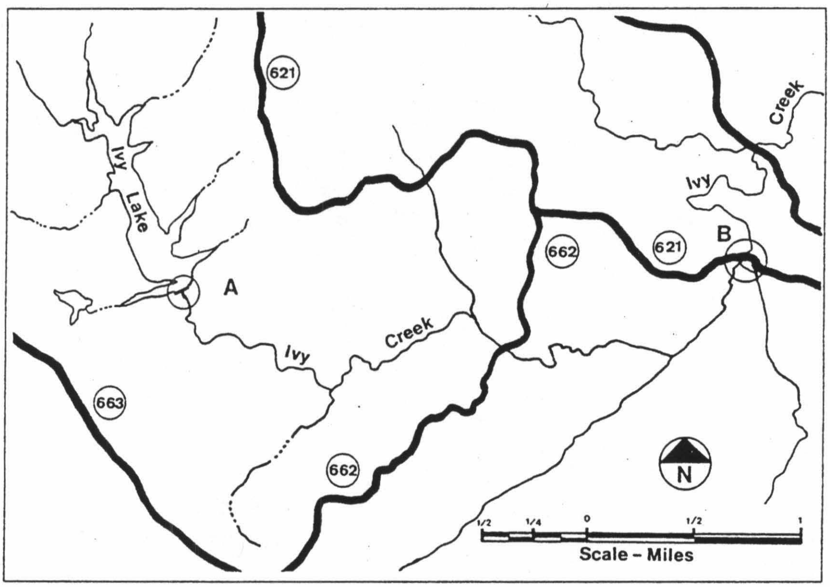
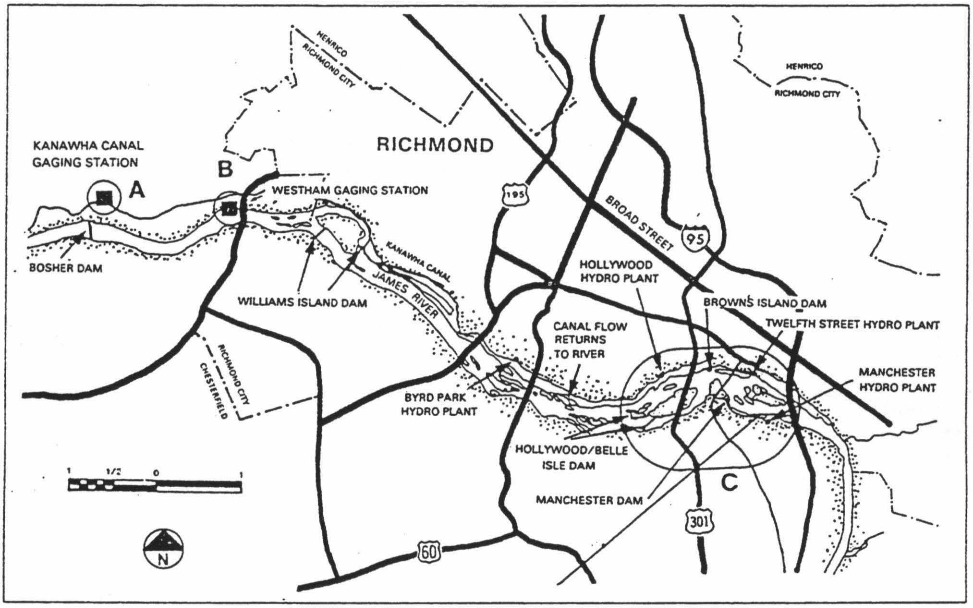


Figure 3. Three study sites (represented as letters A-C) in the main- and side-channel areas of Richmond, in the middle James River watershed (mideastern Virginia). The downstream site (C) was the target for the instream-flow recommendations of Table 3. This map is modified from CDM (1986).



The Virginia Water Resources Research Center is a federal-state organization established at Virginia Polytechnic Institute and State University in 1965 under provisions of the federal Water Resources Research Act of 1964.

Under law, the Center's activities are to:

- consult with the General Assembly, governmental agencies, water user groups, private industry, and other potential users of research;
- establish and administer research agreements with all universities in Virginia;
- facilitate and stimulate research that concerns policy issues facing the General Assembly, supports water resource agencies, and provides organizations with tools to increase effectiveness of water management;
- disseminate new information and facilitate application of new technology;
- serve as a liaison between Virginia and federal research funding agencies as an advocate for Virginia's water research needs; and
- encourage the development of academic programs in water resources management in conjunction with the State Council on Higher Education.

The Water Center is a member of the National Institutes for Water Resources. More information on programs and activities may be obtained by writing or telephoning the Water Center.

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