HABITAT EVALUATION AND PRODUCTION OF ROCK BASS IN TWO VIRGINIA STREAMS

by

Paul Pajak

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APPROVED:

R.J. Neves, Chairman

D.J. Orth

J.J. Ney

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(ABSTRACT)

The use of Habitat Suitability Index (HSI) models for fisheries impact assessment has not been field validated for most fish species. This research sought to test several of the fundamental assumptions inherent in applications of Suitability Index (SI) curves with the Habitat Evaluation Procedures (HEP) of the U.S. Fish and Wildlife Service. Using draft SI's for rock bass (*Ambloplites rupestris*), tests for positive correlations between standing stocks and measures of fish habitat were conducted in 1982-1983. Multiple regression modeling and analysis of fish production were used to further evaluate the efficacy of habitat assessments for rock bass based on HSI models.

Highly significant positive correlations ($p<.01$, $r^2=.68$) between Habitat Units (HU's) and biomass were observed in Back Creek for November 1982. However, significant positive correlations were not observed consistently for other months, models, or streams evaluated. Factors which apparently accounted for low correlations included seasonal fish movements, fish sampling biases, habitat homogeneity, and the omission of

potentially important variables, such as depth and cover, from HSI models. Fish production was essentially the same in Little Walker and Back Creeks and averaged 1.04 $gm^{-2}yr^{-1}$. The factors influencing correlations between fish abundance and habitat measurements prevented meaningful comparisons of rock bass production and the related physical habitat.

Results from this study suggest that summer low flow periods may not always be most limiting to rock bass habitat in Virginia streams. Furthermore, seasonal movements of fish and sampling biases, if not considered, may seriously confound attempts to relate indices of carrying capacity and habitat model outputs. Incorporating the annual home range of the evaluation species into future study designs is recommended to expedite subsequent attempts to validate HSI models.

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INTRODUCTION

Increasing and often conflicting demands on our natural resources underscore the need for more comprehensive and quantitative impact assessment techniques. In fact, NEPA and other environmental legislation explicitly require and are largely responsible for the recent development of habitat-based methodologies (Armantrout 1981; Zagata 1982). However, Zagata (1982) noted that most federal agencies have developed "functional" inventory methodologies, but still lack ecologically-based systems capable of fully meeting their impact assessment mandates. A recent appraisal of public land use policy under the U.S. Department of Interior reports substantive changes in the decision-making process of this agency which favor commercial development, and portends substantial adverse impacts to fish and wildlife habitat (Hair 1982). Such a trend in the national land use policy further obviates the need for impact assessment methods which will fully integrate fish and wildlife resources into allocation decisions.

The fisheries profession has recognized the need for a sound, ecologically-based evaluation procedure in order to better maintain and manage all aquatic resources (Platts 1980). In particular, warmwater stream habitats are most often affected by man's activities because of their close association with good farmland (Funk 1970), yet fish-habitat relationships for most warmwater species are not well understood (Larimore 1981; Stalnaker 1981; Peters 1982). Reasons offered for the relative paucity of data on warmwater systems have included a traditional emphasis on coldwater salmonids (Larimore 1981; Neves 1981), sampling difficulties in warmwater environ- ments related to increased turbidity and fluctuations in discharge (Cleary and Greenbank 1954; Larimore 1961), and the greater species diversity and complexity typical in warmwater streams (Larimore 1981). These reasons should not, however, overshadow the recreational importance of our warmwater streams, which have provided virtually the only form of angling in nearly thirty states (Funk 1970).

In response to this need for fish habitat evaluation, the Fish and Wildlife Service (1980) has developed the Habitat Evaluation Procedured (HEP). HEP is a species-habitat approach to impact assessment which is based on the fundamental assumption that habitat quality and quantity can be numerically described. Habitat "quality" for a single species is determined using Suitability Index (SI) curves developed from the literature for important habitat variables (e.g. temperature, substrate, current velocity). These curves are used to assign values of 0 to 1.0 for each field measurement; 0 indicates unsuitable habitat and

1.0 denotes optimal conditions. The "quantity" of habitat is simply the area over which a particular SI value is assigned. SI values are subsequently combined in components (e.g. reproduction, water quality, food-cover) which are in turn used in mechanistic models to derive an overall Habitat Suitability Index (HSI). It is this latter value which is multiplied by its corresponding area to determine the total number of Habitat Units (HU) available to the evaluation species under the conditions observed. These units, or changes in them, are ultimately used by decision-makers to assess the relative impacts of proposed actions.

The assumption that HSI and carrying capacity are linearly related is critical to the use of HSI with the HEP (USFWS 1980a). This assumption has not been validated for any species to date. Farmer et al. (1982) confirmed the need for validation of land-use planning models and stated that the ultimate objective of this process is to compare model behavior with observed animal abundance. It should be noted, however, that HSI values are not synonymous with the entire HEP management system, which may employ data from any number of sources (USFWS 1980a). Only validation of such decision-making models and methods will ensure widespread, interdisciplinary acceptance, and minimize costly and often counterproductive litigation. Orth and Maughan (1982) recognized this potential problem with the frequent, but unproven application of the Instream Flow Incremental Methodology (IFIM) in many warmwater streams. They concluded that environmental factors controlling the successful reproduction and survival of young black basses (Micropterus spp.) are more complex than the IFIM approach implies, and that further tests of basic assumptions are needed to assess the relative efficacy of that approach. The similar reliance of both methods on SI curves suggests that validation is essential to the overall utility of HEP to fisheries resource managers.

Fish production has traditionally been accepted by many to be an effective measure of a fish population's response to its environment (Waters 1977; Chapman 1978; Neves 1981) and has been considered the "best measure" to assess the overall performance of a particular population (Le Cren 1972). As a single parameter, it embodies the dynamic components of growth, mortality, recruitment, population density and migration (Le Cren 1972). For these reasons, production was selected as a second HEP validation statistic to be used in addition to standing stocks, a relatively static measure by comparison.

This study was undertaken to assess the relative utility of SI curves in determining changes in available habitat (i.e. as measured in Habitat Units) within the HEP framework. The approach was to use prescribed field evaluation techniques with minimal revision and to collect sufficient information to allow for independent evaluation of HEP/HSI assumptions. An effort was made to sample over a range of unsuitable to optimal rock bass habitats to improve the reliability of model evaluation. The rock bass was chosen as the study species because of the availability of SI curves, and the species' importance as a game fish in the Mississippi River drainage. Except for draft SI curves, all tested models were developed in this study and incorporate published guidelines (Terrell et al. 1982) and similar models developed for the green sunfish (Stuber et al. 1982).

Thus, the specific objectives of this study were: 1) to test the assumption that biomass of rock bass and habitat units are positively correlated, 2) to relate standing stocks to select physical and chemical habitat variables and, 3) to estimate annual production of rock bass and relate this value to habitat units.

MATERIALS AND METHODS

Study Areas

Two fourth-order tributaries to the New River were selected for this study using the following criteria: 1) presence of rock bass, 2) minimal fishing pressure, 3) relatively stable and unimpacted watersheds, and 4) year-round accessibility.

Little Walker Creek and Back Creek are located in adjacent watersheds of Pulaski County, Virginia (Figure 1). Upper reaches of Little Walker Creek originate in Bland County and drain the relatively undeveloped deciduous stands of the Jefferson National Forest. Pasture land is common, but remains confined to a narrow strip along the 27.0 km of the fourth-order reaches. The watershed is typically trellised and extends east-northeast until its northward confluence with Walker Creek. In contrast, Back Creek is marked by a dichotomous watershed and much broader floodplain. Land use is predominantly agricultural along the fourth-order reaches, which course easterly for 18.8 km before joining the New River.

Two study sections, each 250 m in length, were established at downstream (I) and upstream (II) locations on both creeks. Lower and upper study sites were located approximately one fourth and three fourths of the distance from the mouth of each stream, respectively. Each study

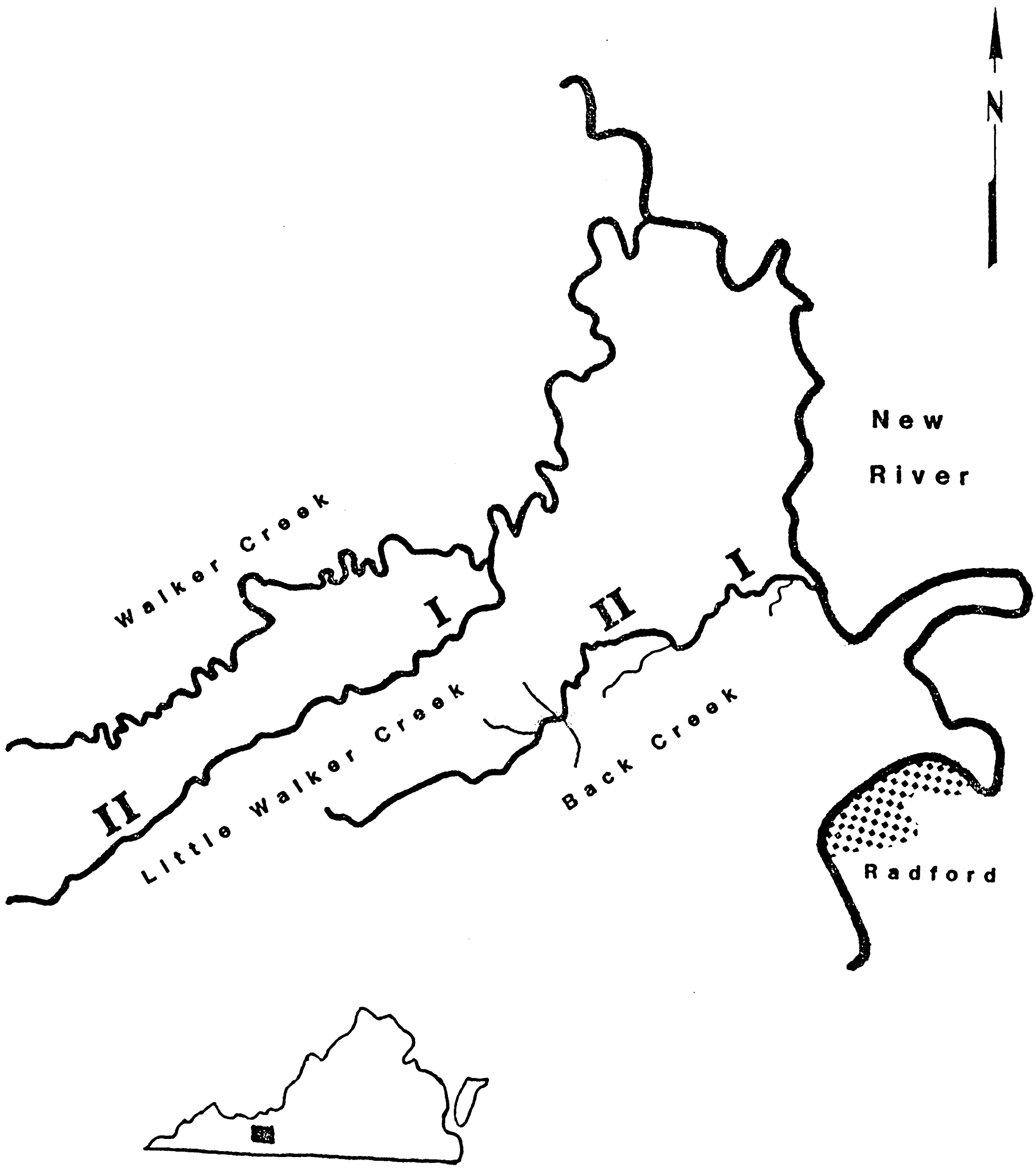


Figure 1. Location of study streams and sections (50 m each) in Pulaski County, Virginia.

section was divided into 50 m subsections that were designated A through E proceeding upstream. The presence of rock bass was confirmed at all sites in early May 1982 using backpack electrofishing gear.

Fishing was evident or reported by landowners at both downstream sites, particularly on Back Creek. Other anthropogenic impacts, potentially limiting stream fish populations, were not observed at any of the sites.

Habitat Measurements

Water quality data were collected in the deepest pools of each 250-m section and were assumed to reflect conditions of the five contiguous 50-m subsections at each of the four study sites, respectively. All measurements were taken on approximately the last day of each month from July 1982 through June 1983. Total alkalinity (mg/l $CaCO_3$), total hardness (mg/l $CaCO_3$), and turbidity (FTU) were measured in the field using a DR-EL/2 Hach Kit (Hach Chemical Company 1977). Conductivity (umhos/cm) was also measured in the field with a Yellow Springs Instrument Model 33 S-C-T portable meter. Minimum, maximum, and ambient water temperatures were recorded using Taylor maximum-minimum thermometers. A Corning Model 12 PH meter was used to measure pH from water samples collected and returned to the laboratory.

All physical habitat measurements were taken during late summer, the assumed low flow period for study streams as evidenced by discharge data from a connected basin (Figure 2). This period was chosen on the assumption that summer low flows minimize available habitat and are thus critical to fish survival (Stalnaker 1981). Physical habitat variables were measured in each 50-m subsection and included stream gradient, canopy cover, mean depth, mean and bottom current velocity, dominant and secondary substrate type, and percent cover.

Stream gradient and canopy cover were evaluated over each 50-m subsection independent of other variable measurements. Gradients were measured with an Abney level and staffs of equal height placed at each end of study subsections. September canopy cover was estimated to the nearest four percent using a hand-held spherical densiometer. The latter measurements were taken at three equally spaced points within each subsection. At each point, percent canopy was estimated at four 90° intervals, the first in the upstream direction.

Remaining microhabitat variables were measured individually within each of approximately fifty cells established in every subsection. These evaluation cells were created by equally spacing transect lines across the stream at intervals of mean stream width. Transect lines were marked in 1-m increments. The net result was a

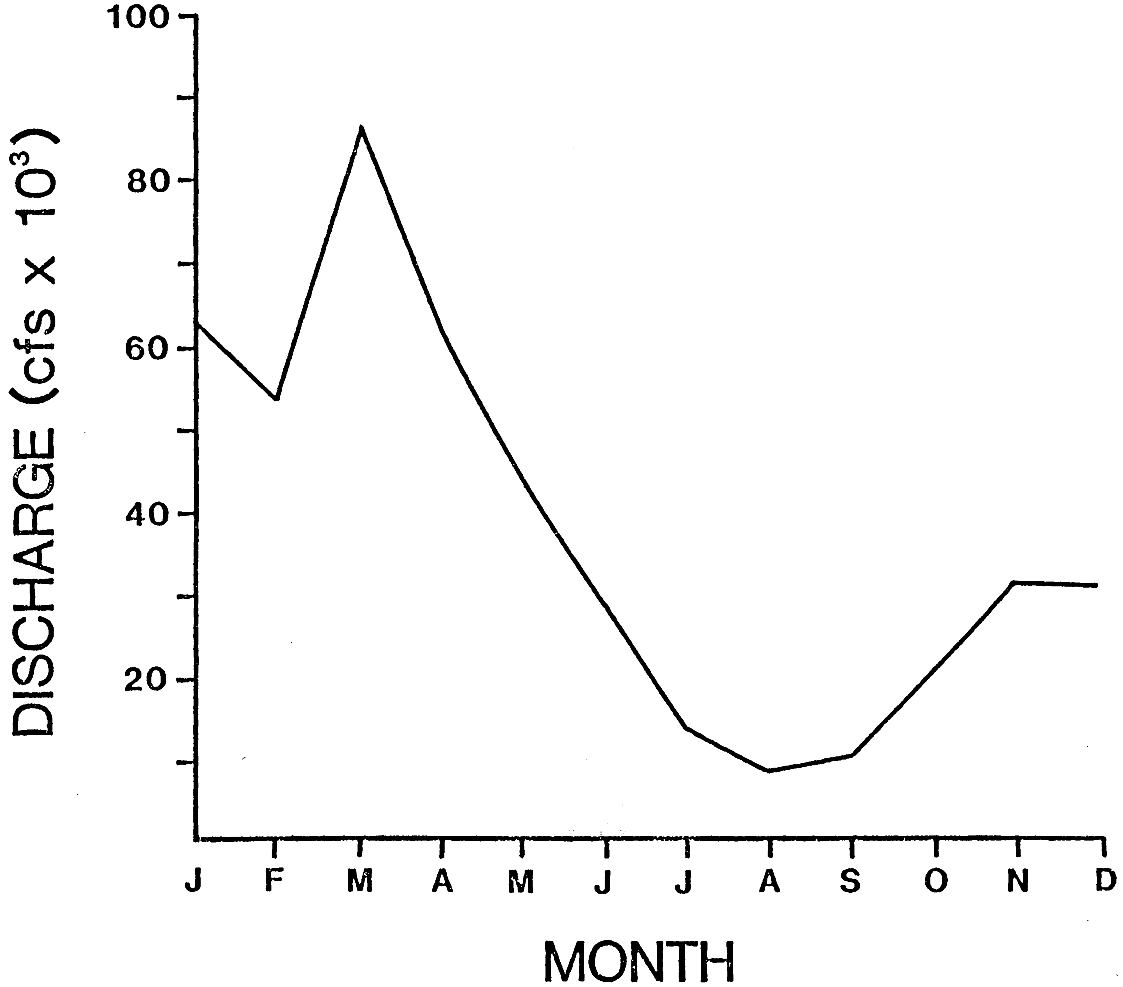


Figure 2. Mean daily discharge by month of Walker Creek at Bane, Virginia for years 1974-1980 (from HISARS 1983).

gridwork of sample cells, each 1 m wide and approximately the mean stream width in length, which encompassed all available habitat for individual subsections.

Within each cell, percent wetted surface area (i.e. portion of the cell within stream margins) and percent cover were visually estimated. No distinction was made between specific cover types such as undercut banks, boulders, or woody debris. Depth (D) was measured to the nearest centimeter in the center of each cell. A Pygmy Gurley meter (Gurley Hydrological Instruments) was used to take current velocities on the bottom and at 0.6 D from the water surface in the center of each cell. Modal substrate size was visually estimated within each cell using a modified Wentworth scale (Table 1). From these data, dominant and secondary substrate types for each subsection were then determined using PROC FREQ (Statistical Analysis System 1982), which computed the number and percent of cells containing each substrate type.

Habitat Suitability Models

Two models were developed in this study to calculate habitat suitability for stream populations of rock bass. Both models were based on an existing riverine model published for green sunfish (Stuber et al. 1982) and employed draft suitability curves for select habitat variables prepared for rock bass (USFWS 1980b). Model I

Table 1. Substrate types, codes, and particle diameter ranges (USFWS 1980b).

Substrate Type	Codes	Particle Diameter (cm)
silt	1	< 0.2
pebble	2	0.2 - 1.5
gravel	3	1.5 - 5.0
cobble	4	5.0 - 20.0
boulder	5	> 20.0
bedrock	6	N/A
rooted vegetation	7	N/A

N/A - not applicable

included all life stages, whereas Model II considered habitat suitability for adult and juvenile life stages only. This second model was formulated after preliminary sampling indicated that only larger size classes of rock bass were fully vulnerable to the electrofishing gear.

In all, four life stages and five variables were evaluated using a total of seventeen available suitability index (SI) curves (Appendix Figure 1). Life stages that were considered were adult, juvenile, larval/fry, and spawning/ embryo. Separate curves were used for each life stage to assign SI values for field measurements of substrate, temperature, current velocity and turbidity. A single curve was used to evaluate suitability of stream gradient for all life stages combined.

Substrate suitability was weighted according to the relative percent of each type present in a given sub-section. Percent substrate type was computed as the number of cells containing a particular type divided by the total number of cells evaluated in the subsection. Each of these percentages was multiplied by the corresponding SI and then summed to determine the weighted suitability for substrate. This procedure was repeated for each life stage and sub-section.

Temperature suitability, for all life stages and subsections, was determined from monthly measurements taken at a single location within each 250-m section. The "maximum temperature" variables (V_5, V_7, V_8) were all equal to the highest stream temperature measured in summer. The "most suitable" temperature for spawning/embryo life stages (V_6) was assumed by interpolation from monthly measurements spanning the observed spawning period in late June.

Turbidity measurements were also taken in only one subsection each at upper and lower stream sites and assumed to be representative of all others within the respective study sections. An annual mean was used to assess adult (V_1) and juvenile (V_4) suitability, the June-July mean for spawning/embryo life stages (V_2), and the July observation for larval/fry stages (V_3).

Current velocity suitability was assigned using subsection means and minimums measured during the September through November evaluation period. "Average current velocity", used with adults (V_9) and juvenile (V_{12}) SI curves, was taken as the mean of all cell measurements within individual subsections. The lowest velocities observed within any cell where depth exceeded 10 cm were used to assign SI values for spawning/embryo (V_{10}) and larval/fry (V_{11}) life stages.

SI values of select variables were then incorporated into mathematical equations of appropriate model components which were subsequently combined into an overall habitat suitability index (HSI) (Figures 3 and 4). Model I

MODEL I

<u>Variables</u>

turbidity: V_1-V_4

temperature: V_5-V_8

Current velocity: V_9-V_{12}

substrate: V_{13}-V_{16}

gradient: V_{17}

<u>Components</u>

Water Quality (C_{WQ}):

$$C_{WQ} = \frac{V_1 + V_3 + V_4 + V_5 + V_7 + V_8}{6}$$

Reproduction (C_R):

$$C_R = (V_2 \times V_6 \times V_{10} \times V_{14})^{1/4}$$

Other (C_{OT}):

$$C_{OT} = \frac{V_{17} + \left(\frac{V_9 + V_{11} + V_{12}}{3}\right)}{2}$$

<u>HSI determination</u>

$$HSI = (C_{WQ} \times C_R \times C_{OT})^{1/3}$$

Figure 3. Variables, model components, and HSI equation for all life stages of rock bass in riverine environments.

MODEL II

Variables

turbidity: V_1-V_4

temperature: V_5-V_8

current velocity: V_9-V_{12}

substrate: V_{13}-V_{16}

gradient: V_{17}

Components

Water Quality (C_{WQ}):

$$C_{WQ} = \frac{V_1 + V_4 + V_5 + V_8}{4}$$

Other (C_{OT}):

$$C_{OT} = \frac{V_{17} + \left(\frac{V_9 + V_{12}}{2} \right)}{2}$$

HSI determination

$$HSI = (C_{WQ} \times C_{OT})^{1/2}$$

Figure 4. Variables, model components, and HSI equation for adult and juvenile life stages of rock bass in riverine environments.

included all four life stages, whereas Model II included adult and juvenile life stages only. To obtain the number of Habitat Units (HU) within each subsection, HSI values were then multiplied by the total area.

Population Estimates and Movements

Rock bass populations were sampled three times in all study subsections during the period from July 1982 to July 1983. The first, or summer sample, was completed in late July and early August, well after the expected spawning period for this species in middle and northern latitudes (Scott and Crossman 1973; Pflieger 1975; Buynak and Mohr 1979; Gross and Nowell 1980). The fall and spring samples were taken in November and the following June, respectively. Early June sampling preceded the observed reproductive period at all study sites and was intended to include the production of gonadal tissue, an important constituent and source of potential bias in estimates of annual production (Chapman 1978, Neves 1981).

Population estimates for each 50-m subsection were determined using the methods of Carle and Strub (1978) with catch data from three consecutive depletion runs of equal fishing effort (Zippen 1958). A Coffelt Model BP-1C backpack electro- shocker was used for all fish sampling. Blocknets, (12-mm square mesh) placed prior to sampling, were used to prevent immigration and emigration from individual subsections during electrofishing efforts. All depletion data were stratified into two size classes prior to population estimation to minimize variability due to gear selectivity (Platts et al. 1983). Individuals less than or equal to 120 mm in length were considered "juveniles"; whereas those longer were termed "adults." This termi- nology was maintained throughout the study. Rock bass were anesthetized using tricaine methanesulfonate (MS-222) before handling and allowed to fully recover prior to release. Individual fish were measured to 1-mm, and weighed to 0.1 g on a Dial-O-Gram triple beam balance (Ohaus Scale Co.). Scale samples were taken according to Lagler (1956), and all specimens later aged from acetate impressions of select scales using a Bausch and Lomb overhead projector.

Net movement of individuals from August to November 1982, was determined from the mark and recapture of rock bass within each subsection. Each fish was given a unique batch mark in conjunction with other biological data collected in August. Adults, as defined earlier, were marked in the soft dorsal fin using the fin-ray scarring method of Welch and Mills (1981). A unique sequence of fin ray clips was applied to all adults within each subsection (Table 2). Juveniles were also marked differently by subsection using subcutaneous dye injections, a technique suited to specimens of small size. A 6.5% Alcian Blue solution was

Table 2. Fin ray scar (adults) and dye (juveniles) marks by subsection in Little Walker and Back Creeks.

Subsection	Juvenile (< 120 mm)	Adult (> 120 mm)
A	LJ - TD	. .
B	RJ - TD	. - .
C	LJ - RJ	. - - .
D	LJ - TV	. - - - .
E	RJ - TV	. - . - .

L	left	T	tail	.	fin ray scar
R	right	D	dorsal	-	unclipped fin ray
J	jaw	V	ventral		

injected under the skin in various combinations of lower jaw and tail locations in order to distinguish among subsections (Table 2). Net distance and direction moved during the four-month period were determined from the recapture of marked individuals in November. Mark retention for both methods was verified under laboratory and field conditions.

A supplemental winter survey for marked rock bass was also conducted in January 1983 to document the occurrence and extent of rock bass movement outside the designated study areas. Prohibitively deep pools and other natural barriers prevented effective surveys at all but section II on Little Walker Creek. At this latter location, approximately 250 m was electrofished above and below the study section. Location of all recaptured rock bass was noted and distance to the subsection of initial capture measured. Depth, current velocity, and dominant substrate were also recorded in areas where marked rock bass were recaptured.

Production

Annual production was estimated using the instantaneous growth method (Ricker 1946) and pooled population data for each of the four 250-m sections on both streams. Ricker's method computes production (P) as the product of the instantaneous growth rate (G) and mean biomass in grams wet weight (B), or P = GB. Instantaneous growth was calculated as $G = \ln W_2 - \ln W_1$, where W_2 is the mean weight at the end of an interval and W_1 is the beginning mean weight for a particular age-class. Mean biomass (B) was equal to the arithmetic mean standing stock for each interval or, $B = (B_1 + B_2)/2$, where point estimates of biomass (B_1, B_2, etc.) were the products of mean individual weight (W) and population number (N). Production was calculated for each cohort, ages 0 to 4, age 5 and older fish combined, and totalled for each 250-m study section.

With the exception of age 0 fish, cohort population estimates were determined from the relative proportion of each age-class observed in the total sample. Initial population size at emergence for age 0 rock bass was estimated using total fecundity (Mathews 1971) and the population sample in June. Fecundity of female rock bass was determined by gravimetric subsampling methods (Bagenal and Braum 1978) using the ovaries of four ripe females ($\bar{W}$ = 126g $\pm$ 32). These data were combined with an assumed age at maturity of three (Carlander 1977) and equal sex ratios for this species (Gross and Nowell 1980). Total fecundity, or initial cohort size, was then calculated for each study section as the product of the number of mature females in the population and their individual fecundity.

Mean weight for age 0 fish at emergence was assumed to be equal to the observed weight of a ripe egg, an assumption typically made for this cohort (Chadwick 1976). This weight was used in calculations of production for the first growth interval in all study sections. Growth rates for subsequent intervals were based on measured mean weights of age 0 fish sampled during this study. Mean weights for all other age-classes were computed using an age and growth computer program (Marques et al. 1982).

Statistical Analysis

Spearman rank correlation coefficients (Hollander and Wolfe 1973) were calculated to test the assumption that subsection HSI's and rock bass standing stocks were positively correlated within each stream (n=10). Similar tests were performed on Habitat Units (HU's) and biomass to examine the sensitivity of correlations to area. Tests were one-sided and only significance levels for positive correlations are reported.

Principal component analysis was used to screen all continuous habitat variables for linear associations and to reduce the number of variables for multiple regression modelling (Hotelling 1933). Variables selected using this procedure were then used with maximum r-square and stepwise multiple regression techniques to develop the best descriptive models relating rock bass standing stocks to select habitat parameters. The best models for each stream were chosen on the basis of model simplicity, r-square, and total squared error (Cp). Mallows (1964) recommended choosing the model where Cp first approaches the number of model parameters. Cross validations of the best models developed for Little Walker Creek were evaluated using data collected on Back Creek. Residuals were calculated as the difference between observed standing stocks in Back Creek subsections and those predicted from the best models developed in Little Walker Creek. An r-square for the tested model was computed as follows (both summer and fall models sampled):

$$r^2 = 1 - \frac{\text{residual sum of squares}}{\text{total sum of squares}}$$

Statistical Analysis System (SAS 1982) programs were used for all frequency, principal component, and multiple regression analyses, and for general summary statistics.

RESULTS

Macrohabitat and Water Quality

Depth and other morphometric characteristics of the four study sites, two each on Little Walker and Back Creeks, were more similar at downstream and upstream sites between streams than within streams (Table 3). Notable exceptions to this trend were consistently higher mean and maximum current velocities and greater canopy cover at both sites on Back Creek. Finer substrates were also more prevalent in Back Creek.

Mean annual water temperatures were higher overall in Back Creek, but maximum temperatures in August were consistently higher at both study sections on Little Walker Creek (Table 4). Freezing temperatures and anchor ice formation did not occur until December in both streams. Although temperature and pH regimes were relatively similar between streams, levels of suspended and dissolved solids were generally much higher in Back Creek as indicated by measures of conductivity, hardness, alkalinity, and turbidity. Cattle grazing and other agricultural land uses were more developed in riparian zones of Back Creek and likely influenced these measurements of suspended and dissolved solids.

Table 3. Habitat characteristics for downstream (I) and upstream (II) study sites on Little Walker and Back Creeks.

Variable	Little Walker Creek I	Little Walker Creek II	Back Creek I	Back Creek II
Surface area (ha)	0.28	0.16	0.24	0.13
Mean depth (m)	0.29	0.15	0.35	0.23
Maximum depth (m)	1.39	0.72	1.03	0.93
Area > 0.25 m depth (ha)	0.14	0.03	0.18	0.05
Mean width (m)	12.40	8.80	10.00	5.40
Mean velocity (cm/sec)	6.00	6.00	8.20	16.00
Maximum velocity (cm/sec)	82.40	35.70	164.80	89.40
Substrate:				
Primary (%)	cobble (57)	cobble (57)	silt/sand (76)	bedrock (32)
Secondary (%)	gravel (14)	boulder (33)	cobble (17)	silt/sand (30)
Cover (%)	8.7	13.3	15.8	3.2
Canopy (%)	24.2	24.7	89.9	36.3

Table 4. Annual means and ranges (in parentheses) of select water quality parameters in Little Walker (LWC) and Back Creeks (BC) from July 1982 through June 1983.

Variable	Stream Section			
	LWC I	LWC II	BC I	BC II
Temperature (°C)	13.9 (-1-29)	11.8 (0-27)	13.1 (-1-26)	12.8 (0-25)
Conductivity (umhos/cm)	40.7 (23-72)	43.2 (23-80)	298.6 (205-435)	304.8 (210-445)
Hardness (mg/l $CaCO_3$)	21.8 (10-35)	23.5 (13-39)	231.4 (169-280)	230.7 (175-284)
Alkalinity (mg/l $CaCO_3$)	22.3 (10-35)	20.8 (10-35)	212.9 (165-250)	213.3 (170-260)
Turbidity (FTU)	3.6 (0-10)	4.5 (0-8)	10.9 (3-26)	14.2 (9-30)
pH	7.8 (7.2-8.3)	8.0 (7.7-8.1)	8.1 (7.7-8.5)	7.9 (7.4-8.1)

Discharge during the study year, 1982, was probably much lower than average according to hydrologic data available for Walker Creek, a tributary to the New River into which Little Walker Creek flows (Figure 5). Although 1982 appeared to be a drier than average year, the observed low flow period in August/September was consistent with average flow patterns.

Habitat Suitability

Differences in Habitat Units (HU) between subsections explained as much as 68 percent of the variation in total rock bass biomass (Table 5). Coefficients of determination (r^2) and associated significance levels were higher in November than in any other sample and for all three samples combined. No significant correlations using either model were observed for June. HU's computed using Model I were consistently better correlated with biomass than those from Model II. Overall, HU's and rock bass biomass showed a greater correlation in Back Creek than in Little Walker Creek.

Similar analysis of Habitat Suitability Index (HSI) values and rock bass standing stocks (g/m^2) yielded lower and less significant r^2 values for the positive correlations observed (Table 6). The only highly significant ($P<.05$) correlations were for November data on Back Creek. The absence of any correlation for November on

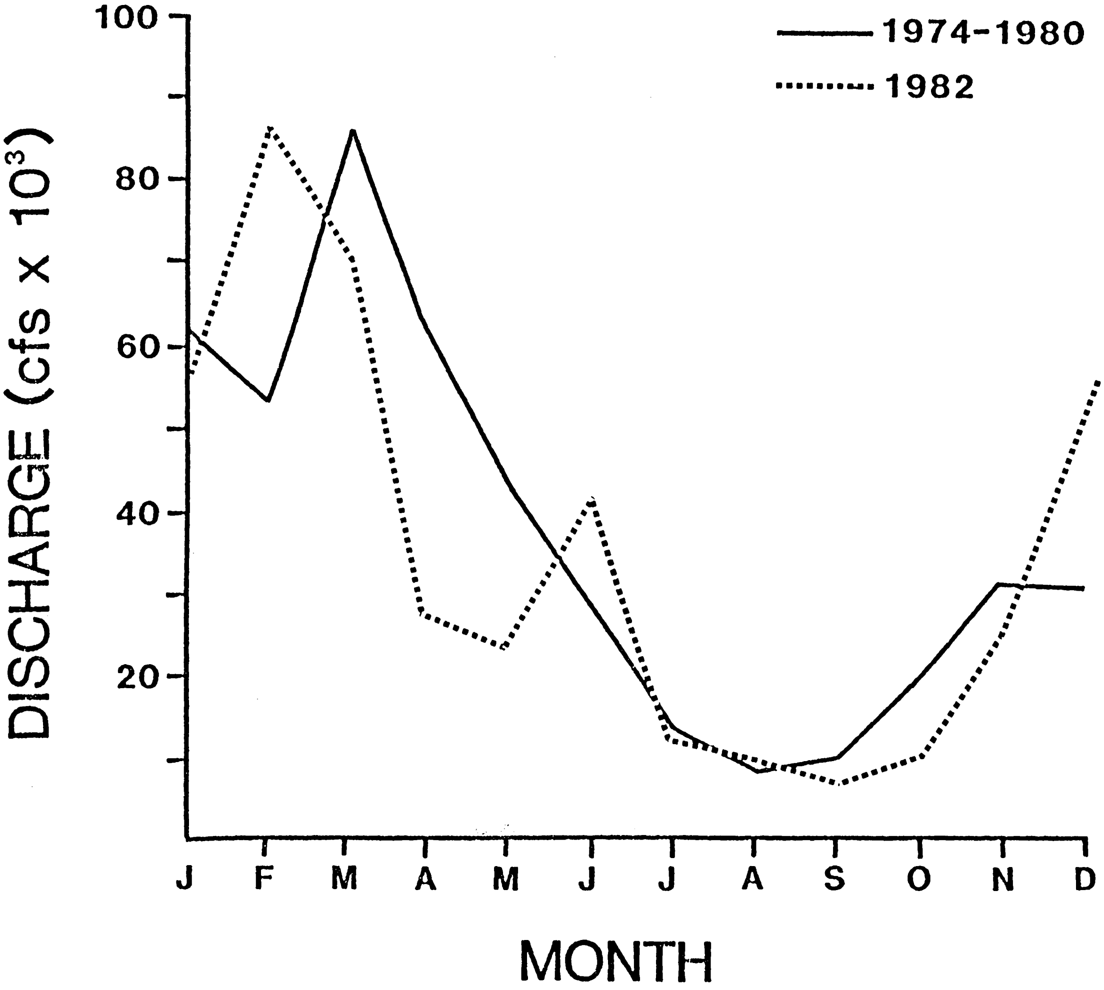


Figure 5. Mean daily discharge by month of Walker Creek for study year (1982) and a seven year average (1974-1980), (from HISARS 1983).

Table 5. Coefficients of determination (r2) and significance levels (in parentheses) of Habitat Units and rock bass biomass (g) in Little Walker and Back Creeks.

Location	Model	Month: Aug	Nov	Jun	combined
LWC	I	- (NS)[a]	0.51 (0.02)	- (NS)	- (NS)
	II	- (NS)	0.33 (0.04)	- (NS)	- (NS)
BC	I	0.42 (0.03)	0.68 (0.01)	- (NS)	0.44 (0.02)
	II	0.38 (0.03)	0.64 (0.01)	- (NS)	0.44 (0.02)

[a]Negative correlations considered "not significant" under null hypothesis.

Table 6. Coefficients of determination (r2) and significance levels (in parentheses) of Habitat Suitability Index (HSI) values and rock bass standing stocks (g/m2) in Little Walker and Back Creeks.

Location	Model	Month: Aug	Nov	Jun	combined
LWC	I	0.03 (0.31)	- (NS)[a]	0.13 (0.14)	0.04 (0.28)
	II	0.07 (0.21)	- (NS)	0.19 (0.09)	0.09 (0.09)
BC	I	0.04 (0.27)	0.61 (0.01)	- (NS)	0.22 (0.08)
	II	0.02 (0.32)	0.50 (0.02)	- (NS)	0.25 (0.07)

[a]Negative correlations considered "not significant" under null hypothesis.

Little Walker Creek is in direct contrast with the results for HU's and biomass previously reported (Table 5). For these latter data, November was the only month for which HU's and biomass were positively correlated. It is important to remember that HU's differ only from HSI's by a factor of area (i.e. HU = HSI x area). Therefore the inconsistencies between test results reported in Tables 5 and 6 may be artifacts of variability introduced by the area factor and its apparent influence on the correlation analyses. Relative performance of Models I and II was also inconsistent between streams using HSI's and rock bass standing stocks. More variation in standing stocks was explained in Little Walker Creek using Model II HSI's, whereas Model I correlations were somewhat higher in Back Creek. Considering the relatively low r^2 values and significance levels for most HSI/standing stock data, these inconsistencies in model performance may reflect relatively homogeneous data.

When results were stratified by stream section, HU's using both models correctly ranked mean and summer biomass for three out of four study sections (Table 7). The section incorrectly ranked, LWC I, had the highest number of HU's but ranked only third by biomass estimates. Relative ranking accuracy was similar using fall estimates. However, overall biomass ranks were reversed for the two LWC sections. This latter phenomenon was the

Table 7. Seasonal biomass (g/section) and summer Habitat Units for rock bass in downstream (I) and upstream (II) sections on Little Walker (LWC) and Back Creeks (BC). (Section ranks in parentheses.)

Variable	Stream Section			
	BC I	LWC II	LWC I	BC II
Biomass				
summer	6531 (1)	6451 (2)	3678 (3)	2524 (4)
fall	7194 (1)	793 (3)	3212 (2)	93 (4)
spring	2487 (3)	6937 (1)	3883 (2)	2360 (4)
mean	5404 (1)	4727 (2)	3591 (3)	1659 (4)
Habitat Units				
model I	2340 (2)	1596 (3)	2529 (1)	1202 (4)
model II	2315 (2)	1612 (3)	2390 (1)	1162 (4)

all life stages
adult and juvenile life stages only

result of substantial declines in fall biomass in LWC II. No consistent trend in HU's and spring biomass ranks was apparent. To some extent, the low number of sections confounded comparisons of HU and biomass ranks in all seasons. Although Model II resulted in lower numbers of HU's in all sections, it did not change their relative ranking.

Similar comparisons using summer HSI's from Model I correctly ranked mean annual standing stocks in all four study sections (Table 8). Model I HSI ranks were not in total agreement with standing stock ranks for any of the individual seasons evaluated. Model II HSI's correctly ranked standing stocks in all four sections for summer only. Absolute ranks for Model II HSI's reversed the order of BC II and LWC I to third and fourth, respectively. The only component SI's to correctly rank standing stocks of all sections were those of the "other" component (gradient and current velocity parameters) in Model II. SI ranks for this model component were in total agreement with mean standing stocks of all sections. Deletion of earlier life stages from Model II resulted in general declines in section HSI values with the exception of LWC II. Other effects of Model II were slight increases in SI values for the "water quality" component, total elimination of the "reproduction" component, and consistent decreases in the SI's of the "other" component. The greatest declines were observed in section BC II.

Table 8. Seasonal standing stocks (g/m^2), summer Habitat Suitability Index values, and model component suitability indices for downstream (I) and upstream (II) sections on Little Walker (LWC) and Back Creeks (section ranks in parentheses).

Variable	Stream Section			
	BC I	LWC II	LWC I	BC II
Standing stocks				
summer	2.68 (2)	3.92 (1)	1.32 (4)	1.89 (3)
fall	2.95 (1)	0.48 (3)	1.12 (2)	0.07 (4)
spring	1.02 (4)	4.22 (1)	1.40 (3)	1.77 (2)
mean	2.22 (1)	2.87 (2)	1.29 (3)	1.24 (4)
HSI				
model I	0.96 (2)	0.97 (1)	0.91 (3)	0.90 (4)
model II	0.95 (2)	0.98 (1)	0.86 (4)	0.87 (3)
Component SI				
Water quality				
model I	0.99 (1)	0.98 (2)	0.88 (4)	0.94 (3)
model II	0.98 (2)	1.00 (1)	0.89 (4)	0.96 (3)
Reproduction				
model I	0.95 (1)	0.94 (2)	0.94 (2)	0.90 (3)
model II	-	-	-	-
Other				
model I	0.95 (2)	0.97 (1)	0.89 (4)	0.90 (3)
model II	0.92 (2)	0.96 (1)	0.83 (3)	0.78 (4)

All life stages.

Adult and juvenile life stages only.

Individual HSI values for Models I and II indicated near optimal habitat in most subsections of both study streams (Tables 9 and 10). Subsection HSI's derived using Model I ranged from 0.87 in BC IIE to 0.98 in several subsections of LWC II and BC I. As many as six subsections were identically ranked using Model I HSI's with discrimination between subsections limited to a total of nine unique HSI values. The "reproduction" component most frequently had the lowest SI for Model I followed by the "other" and "water quality" component SI's, respectively.

Model II resulted in the deletion of the reproduction component from computations of HSI's and a slight decrease in expected habitat suitability in all subsections of LWC I and BC II (Table 10). Other noticeable effects of this revised model included small increases in "water quality" component SI's and decreases in "other" component SI's for most subsections. Modified HSI's ranged from 0.80 to 1.00 and increased subsection discrimination to fourteen unique values. The "other" component most often had the lowest SI, particularly in sections LWC I and BC II. Because water quality measurements were taken at only one location at each study section, and assumed to be equal for all corresponding subsections, the water quality component provided no discrimination in habitat suitability at the subsection level.

Table 9. Model I suitability index values, overall habitat quality (HSI), and relative ranks by subsection for Little Walker (LWC) and Back Creeks (BC).

Stream	Section	Sub-Section	Model I Component: Water Quality	Model I Component: Reproduction	Model I Component: Other	HSI	Total Rank
LWC	I	A	0.88	0.91	0.85	0.88	8
		B	0.88	0.93	0.88	0.90	6
		C	0.88	0.96	0.83	0.89	7
		D	0.88	0.96	0.89	0.91	5
		E	0.88	0.96	1.00	0.95	2
	II	A	0.98	0.95	1.00	0.98	1
		B	0.98	0.93	0.93	0.95	2
		C	0.98	0.93	0.93	0.95	2
		D	0.98	0.95	1.00	0.98	1
		E	0.98	0.95	1.00	0.98	1
BC	I	A	0.99	0.95	1.00	0.98	1
		B	0.99	0.95	1.00	0.98	1
		C	0.99	0.95	1.00	0.98	1
		D	0.99	0.95	0.82	0.92	4
		E	0.99	0.95	0.91	0.95	2
	II	A	0.94	0.95	0.93	0.94	3
		B	0.94	0.95	0.78	0.89	7
		C	0.94	0.89	0.87	0.90	6
		D	0.94	0.89	0.88	0.90	6
		E	0.94	0.84	0.82	0.87	9

Table 10. Model II suitability index values, overall habitat quality (HSI), and relative ranks by subsection for Little Walker (LWC) and Back Creeks (BC).

			Model II Component				
Stream	Section	Sub-Section	Water Quality	Repro-duction	Other	HSI	Total Rank
LWC	I	A	0.89	-	0.77	0.83	12
		B	0.89	-	0.81	0.85	10
		C	0.89	-	0.75	0.82	13
		D	0.89	-	0.84	0.86	9
		E	0.89	-	1.00	0.94	4
	II	A	1.00	-	1.00	1.00	1
		B	1.00	-	0.90	0.95	3
		C	1.00	-	0.90	0.95	3
		D	1.00	-	1.00	1.00	1
		E	1.00	-	1.00	1.00	1
BC	I	A	0.98	-	1.00	0.99	2
		B	0.98	-	1.00	0.99	2
		C	0.98	-	1.00	0.99	2
		D	0.98	-	0.73	0.85	10
		E	0.98	-	0.86	0.92	6
	II	A	0.96	-	0.90	0.93	5
		B	0.96	-	0.67	0.80	14
		C	0.96	-	0.80	0.88	8
		D	0.96	-	0.82	0.89	7
		E	0.96	-	0.73	0.84	11

The "reproduction" component, most often limiting in Model I calculations of HSI's, incorporated turbidity (V_2), temperature (V_6), current velocity (V_{10}), and substrate (V_{14}) variables for spawning/embryo life stages. Only Model I utilized a substrate variable, and this was limited to substrate suitability for the spawning/embryo life stage (V_{14}). Of these four variables, substrate measures accounted for the lowest SI values and varied the most between subsections in both streams (Appendix Tables 1-4). The other three variables, all related to water quality, were constant across subsections at each of the four study sites and were optimal (1.00), with the exception of turbidity (V_2) at BC II.

Model II included variables for adult and juvenile life stages only and resulted in the deletion of the "reproduction" component from subsequent HSI calculations (Table 10). With these revisions, the "other" component became limiting in model estimates of habitat suitability. This component included variables for gradient (V_{17}) and current velocity (V_9, V_{12}). SI values for gradient were optimal (1.00) for all subsections and therefore were not limiting, nor did they discriminate between subsections (Appendix Tables 1, 2, 3, and 4). Of the other two variables, average current velocity for juveniles (V_{12}) had consistently lower SI's than those for adults (V_9) in all subsections and ultimately reduced HSI values computed using the second model.

Habitat Utilization

Habitat use by rock bass, particularly adults, varied considerably among seasons. Few stream sections were observed to support consistently high fish densities during the three evaluation periods. The suggested trend in habitat use was to occupy pools and runs during the summer, emigrate to the deepest pools in winter, and recolonize shallow riffles and runs prior to spawning in early summer. Similar, but less pronounced utilization patterns were observed for juvenile rock bass.

Summer habitat supporting the highest rock bass densities was characterized by mean depths ranging from 20-39 cm, current velocities of 10-19 cm/sec, and silt as the dominant substrate type (Table 11). Rock bass densities tended to increase with increasing amounts of cover, the highest densities observed in subsections containing 20-24% cover. When life stages were considered separately, adults were observed to occupy subsections with shallower mean depth, slightly higher current velocities, and coarser substrate types (Table 12). Adults also utilized areas with less cover. The latter observations resulted in a downward shift in the optimal values of all habitat variables with the exception of mean depth. This phenomenon is likely the result of using variable ranges (Table 11) for life stages combined, while variable means were used in the stratified analysis comparing habitat

Table 11. Summer, fall and spring population estimates and densities for rock bass in relation to physical habitat variables in Little Walker and Back Creeks combined; 1982-83.

Variable and Interval	Area Sampled (m^2)	August 1982 (N)	August 1982 (N/m^2)	November 1982 (N)	November 1982 (N/m^2)	June 1983 (N)	June 1983 (N/m^2)	Combined 1982-83 (N)	Combined 1982-83 (N/m^2)
Mean depth (cm)									
0-9	0	-	-	-	-	-	-	-	-
10-19	2985	109	0.037	33	0.011	100	0.034	242	0.081
20-29	2237	169	0.076	58	0.026	119	0.053	346	0.155
30-39	1854	138	0.074	71	0.038	41	0.022	250	0.135
40-49	0	-	-	-	-	-	-	-	-
50-59	1120	46	0.041	54	0.048	27	0.024	127	0.113
Total	8196	462		216		287		965	
Mean velocity (cm/sec)									
0-4	3961	186	0.047	152	0.038	118	0.030	456	0.115
5-9	1831	106	0.058	31	0.017	63	0.034	200	0.109
10-14	1313	96	0.073	15	0.011	61	0.046	172	0.131
15-19	869	64	0.074	18	0.021	34	0.039	116	0.133
20-24	222	10	0.045	0	0.000	11	0.050	21	0.095
Total	8196	462		216		287		965	

Table 11. (continued) Summer, fall and spring population estimates and densities for rock bass in relation to physical habitat variables in Little Walker and Back Creeks combined; 1982-83.

Variable and Interval	Area Sampled (m^2)	August 1982		November 1982		June 1983		Combined 1982-83	
		(N)	(N/m^2)	(N)	(N/m^2)	(N)	(N/m^2)	(N)	(N/m^2)
Substrate									
silt	2698	174	0.064	111	0.041	55	0.020	340	0.126
pebble	0	-	-	-	-	-	-	-	-
gravel	0	-	-	-	-	-	-	-	-
cobble	4183	224	0.054	96	0.023	170	0.041	490	0.117
boulder	591	30	0.051	0	0.000	27	0.046	57	0.096
bedrock	724	34	0.047	9	0.012	35	0.048	78	0.108
Total	8196	462		216		287		965	
Cover (%)									
0-4	2148	74	0.034	27	0.013	67	0.031	168	0.078
5-9	1374	51	0.037	34	0.025	42	0.031	127	0.092
10-14	2731	197	0.072	98	0.036	95	0.035	390	0.143
15-19	753	47	0.062	4	0.005	30	0.040	81	0.108
20-24	258	31	0.120	6	0.023	22	0.085	59	0.229
25-29	932	62	0.067	47	0.050	31	0.033	140	0.150
Total	8196	462		216		287		965	

Table 12. Habitat characteristics of subsections with the highest rock bass densities (upper 25th percentile) by season and life stage.

Habitat variable	Summer		Fall		Spring	
	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Density (N/m2)	.073	.042	.044	.016	.064	.006
Depth (cm)	24.0	33.1	35.1	32.3	21.2	20.4
Velocity (cm/sec)	8.8	8.1	3.2	6.2	11.6	7.6
Cover (%)	15	18	15	13	13	11
Substrate	cobble	silt	silt	silt	cobble	cobble

utilization for individual life stages in subsections with the highest densities of adults and juveniles, respectively (Table 12).

Fall densities of rock bass indicated a distinct shift in habitat use from that observed during the summer evaluation period (Table 11). Overall declines in fish abundance, consistent in subsequent spring population estimates, suggested mass emigration from habitat types present in study reaches. Of the fish remaining, densities were the highest in the deepest subsections sampled. Other characteristics of the most utilized subsections included minimal current velocities (0-4 cm/sec), predominantly silt substrates, and maximum available cover (25-29%). This general shift in habitat use from summer to fall more closely approximated adult utilization patterns than those of juvenile rock bass (Table 12). The smaller size classes appeared to remain in subsections similar to those which they occupied in the summer, except that current velocities and percent cover were slightly lower in fall in subsections with the highest juvenile densities. It should be noted that juvenile densities for many fall estimates were based on a single individual (Appendix Tables 5-6) and may not provide a reliable index of habitat utilization for these smaller size classes.

Spring densities of rock bass were highest in shallower subsections adjacent to wintering pools and containing coarser substrates. These subsections, when compared to those most used in the summer and fall, were generally shallower, had the swiftest currents, and the coarsest substrate types (Table 12). Use of cover was similar to that observed in other seasons; the highest rock bass densities were observed in those subsections with more cover. Except for adult habitat, characteristics were very similar to those of juveniles during the spring evaluation period.

Of those subsections with the highest densities of rock bass (upper 25th percentile) in each season, none were observed to support high densities through all three seasons evaluated. Only four of the twenty subsections had rock bass densities in the upper 25th percentile in more than one season. Two subsections in Little Walker Creek were among those with the highest summer and spring densities, while two subsections in Back Creek had densities in the upper 25th percentile during both summer and fall stock assessments. All four subsections had similar mean depths, current velocities, percent cover, and substrate types, but generally differed in their proximity to other habitat types. The two subsections in Little Walker Creek with high summer/spring densities were riffle-run sequences with predominantly cobble substrates grading into gravel. The two subsections in Back Creek with relatively high summer/fall densities were similar in their general habitat characteristics except that they were contiguous, deeper, slow-moving pools with silt substrates.

Multivariate Analysis

Only continuous variables which were measured separately within each stream subsection were considered for inclusion in multiple regression models. Potential variables included mean current velocity (X1); mean depth (X2); percent instream cover (X3); percent canopy cover (X4); area deeper than 10 cm (X5), 25 cm (X6), or 50 cm (X7), and mean current velocity on the bottom (X8).

Principal component analysis resulted in the selection of the following variables for inclusion in multiple regression models:

X1 = mean current velocity
or
X2 = mean depth
X3 = percent cover
X4 = percent canopy
X5 = area more than 10 cm deep
or
X6 = area more than 25 cm deep
or
X7 = area more than 50 cm deep

Correlated variables are separated by "or" and were not evaluated simultaneously in regression modeling. Mean current velocity (X1) and bottom current velocity (X8) were highly correlated (r=.98). The latter was subsequently deleted from evaluation to minimize variable redundancy and conserve degrees of freedom.

Depth, particularly area more than 25 cm deep (X6), was the most important variable in multiple regression models explaining differences in standing stocks of rock bass between subsections (Table 13). Linear models using only variable X6 accounted for 91 and 89 percent of the variation (r^2) in fall standing stocks in Back and Little Walker Creeks, respectively. Other variables were relatively unimportant and generally accounted for less than 20 percent of the variation in standing stocks.

Collective evaluation of all regression statistics indicated that the "best" models relating standing stocks and physical habitat variables were those for November. In addition to the highest r^2 values and significance levels (P-value), C(P) statistics were nearest the number of model variables for November regressions. Coefficients for the most important variable, X6, were also consistently positive and made more ecological sense than negative coefficients observed for both creeks in July and August. However, relatively high y-intercepts for all models indicate rock bass standing stocks in the absence of habitat (i.e. depth and other variables) and suggest that model predictions would likely be inaccurate.

Cross validation of Little Walker Creek regression models was attempted using Back Creek data. The

Table 13. One-, two-, and three-variable regression models relating rock bass standing stocks (Y) to select physical habitat variables (see definitions below).

Location	Month	Regression Model	r^2	C(P)	P-value
LWC	July	Y = 1587.17 - 1.58(X5)	0.30	6.65	0.10
		Y = 1.001.84 - 1.59(X5) + 12.82(X3)	0.53	4.45	0.07
		Y = 304.65 - 1.93(X6) + 20.33(X3) + 28.77(X4)	0.72	3.24	0.04
	November	Y = 48.09 + 2.12(X6)	0.95	3.16	0.01
		Y = 197.91 + 2.96(X6) - 13.13(X2)	0.95	3.16	0.01
		Y = 137.53 + 3.03(X6) - 15.24(X2) + 2.08(X3)	0.96	3.31	0.01
BC	August	Y = 176.31 + 3.02(X5)	0.47	2.17	0.03
		Y = 120.99 + 6.05(X5) - 47.51(X2)	0.69	1.29	0.02
		Y = 1259.07 - 15.66(X7) + 84.79(X2) + 15.39(X3)	0.76	3.27	0.03
	November	Y = 502.86 + 5.45(X6)	0.91	1.09	0.01
		Y = 271.65 + 7.83(X6) - 44.93(X2)	0.95	2.48	0.01
		Y = 480.58 + 7.59(X6) - 56.14(X2) + 4.02(X3)	0.96	3.53	0.01

X1 = mean current velocity
X2 = mean depth
X3 = percent cover
X4 = percent canopy
X5 = area > 10 cm deep
X6 = area > 25 cm deep
X7 = area > 50 cm deep

three-variable model developed for July standing stocks in Little Walker Creek did not explain any of the variation ($r^2 = 0$) in Back Creek standing stocks for the same period. In contrast, the November model ($Y = 48.09 + 2.12 X6$) from Little Walker Creek accounted for approximately half of the variation ($r^2 = .52$) in Back Creek standing stocks for the same month.

Mark Retention

Subcutaneous injections of Alcian Blue were clearly visible in laboratory specimens after 67 days. Further observation and evaluation of this technique under controlled conditions was prevented by complete mortality of these individuals resulting from unregulated aquaria temperatures.

Of the experimental mark locations tested, lower jaw and tail locations exhibited the best retention. Lower jaw marks were most easily applied and readily visible because of the high contrast with naturally light coloration in this area. Tail locations also retained injections well, but were somewhat more difficult to apply in this more heavily scaled region. Increased pigmentation in this area reduced mark contrast as well. Opercle injections proved the least satisfactory. Dye marks in these areas were uncontrollably diffuse and the most difficult to apply consistently. For these reasons, and for concern over behavior modification due to color pattern alteration in this important area, the opercle was not used for field marking.

Alcian Blue injections showed some fading but remained distinguishable on most recaptured field specimens for periods exceeding seven months (229 days). Observed differences in mark visibility were probably attributable to variable application during initial marking. Lower jaw marks were most visible in field specimens and were usually the first of the paired marks to be observed. Visibility of tail marks was best on blanched or lightly pigmented individuals but identification of darkly pigmented rock bass similarly marked was not apparently reduced. Inconsistent recording of double marked fish precluded reliable estimates of dye mark retention or, more precisely, observability.

Fin ray scars were also highly visible in field specimens after 229 days. No natural scars, potentially confused with applied marks, were observed during initial marking periods. However, two minor problems with this technique were observed in its application to rock bass. First, the clipping of adjacent fin rays occasionally resulted in the loss of the distal portion of these rays and the interconnecting fin membrane. Incidence of this was very low, however, and did not appear to reduce accurate recording of rock bass marked this way. The second difficulty was related to applying this marking technique to smaller specimens (<120 mm). Individuals within this size range were generally too small to mark easily without inadvertent damage to other fin rays.

Overall, both marks were considered "permanent" during the period between samples, and no apparent differences in their discernability were noted. For the purpose of movement analysis, it was assumed that both types were observed and recorded with equal consistency in the November recapture sample.

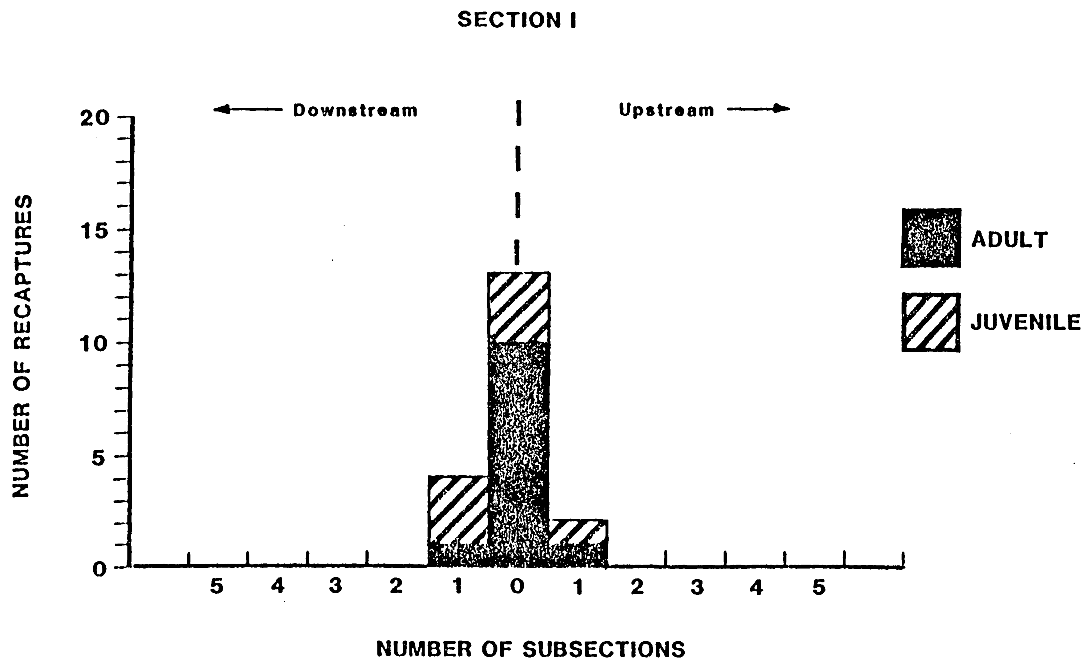
Seasonal Movement

Determinations of net movement from July to November 1982 were based on the recapture of 47 marked rock bass (Table 14). These individuals comprised 11.4 percent of the 411 fish marked in July, and 22.8 percent of the 206 fish captured in November. Overall, recapture data show movement of adult populations into deeper, downstream pools whereas substantial emigration from shallower, upstream study areas was observed for both life stages during this period.

Rock bass downstream remained relatively sedentary from the July to November sampling periods (Figures 6 and 7). The proportion of the total July sample collected in November (C/M) was .71 and .64 for LWC I and BC I, respectively (Table 14). No net movement was observed for

Table 14. Number (R) and percent recapture (R/M) in November of adult and juvenile rock bass by study site in Little Walker and Back Creeks. Number of fish marked in August (M) and total sample size in November (C) are included.

Study	Site	Life Stage	Number Marked (M)	Recapture Sample (C)	Marks Recaptured (R)	% Recapture (R/M)	Total
LWC	I	A	34	36	12	.35	.19
		J	67	36	7	.10	
	II	A	49	7	4	.08	.06
		J	35	9	1	.03	
BC	I	A	59	75	13	.22	.12
		J	118	38	9	.08	
	II	A	17	0	0	.00	.02
		J	32	5	1	.03	
Totals			411	206	47		



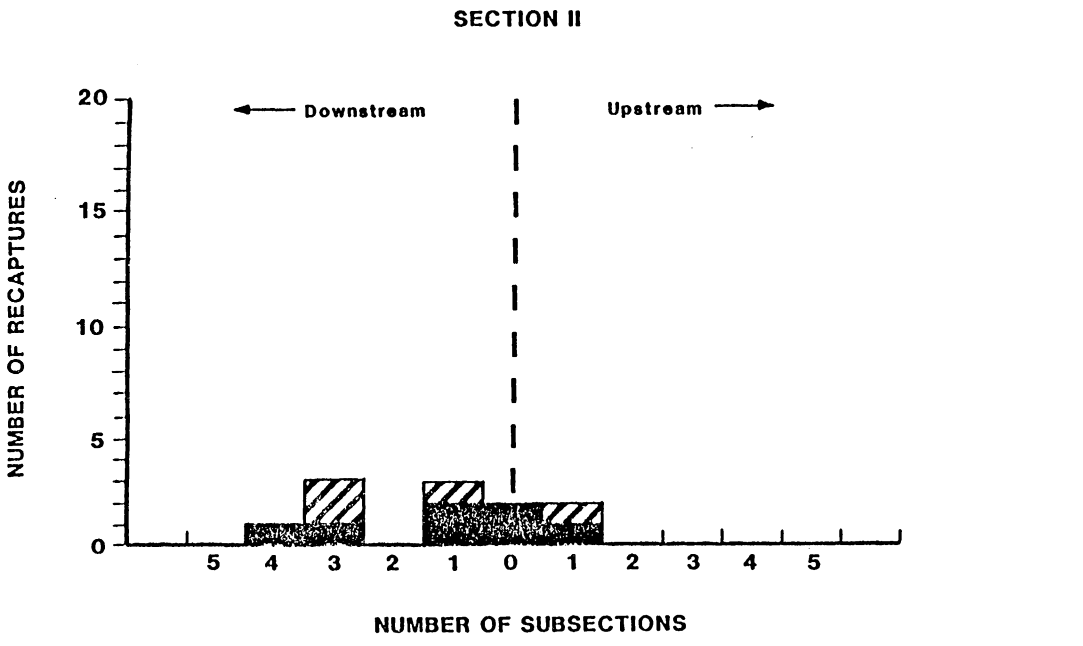


Figure 6. Net movement of adult and juvenile rock bass from July to November in Little Walker Creek. (Site II includes six recaptures from the winter habitat survey, January 1983).

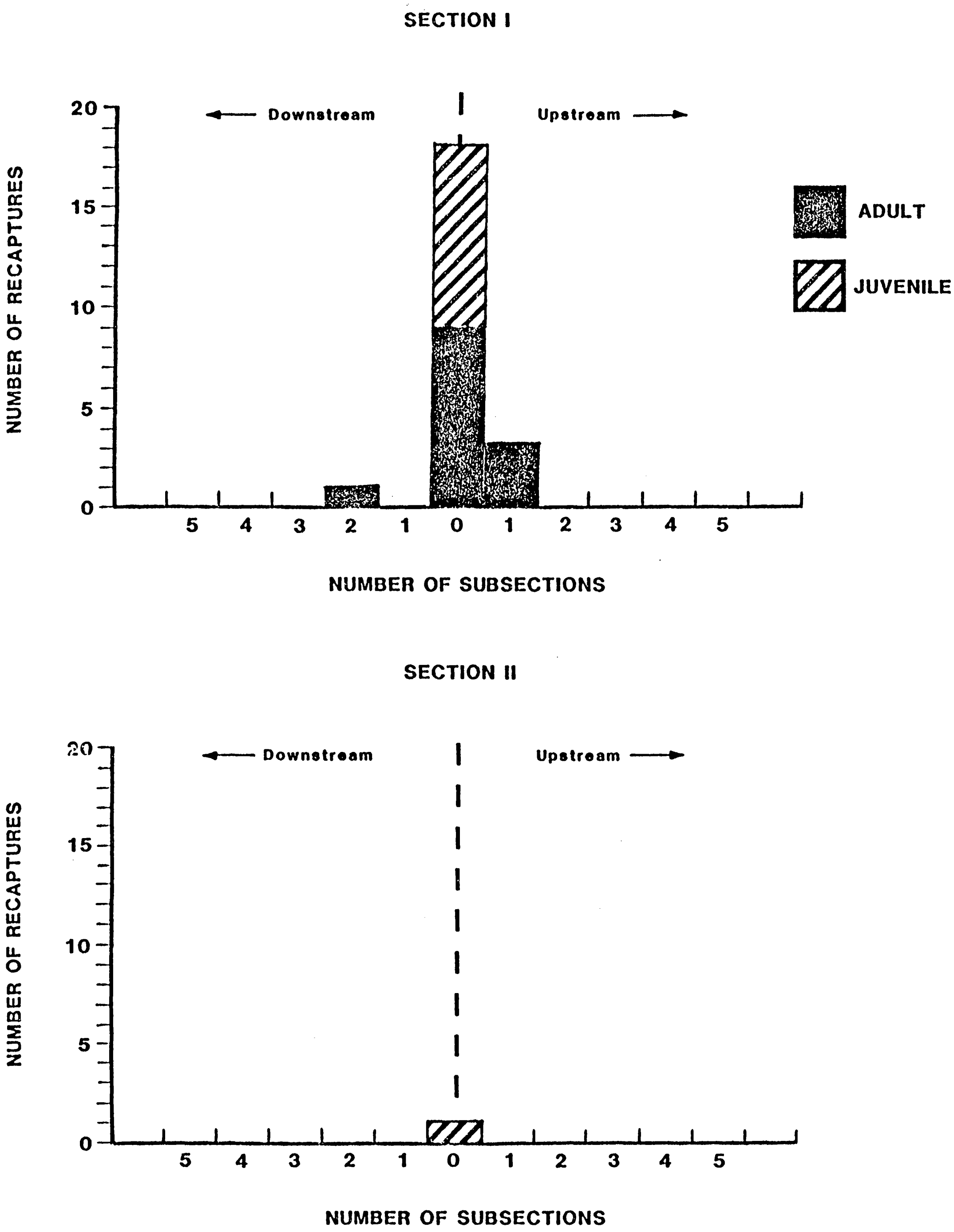


Figure 7. Net movement of adult and juvenile rock bass from July to November in Back Creek.

68 and 82 percent of the recaptured individuals in these same respective sections. Total percent recapture (R/M) was also consistently higher, up to three times greater for downstream versus upstream study sites. Recaptures within any one subsection, however, did not exceed five and were too similar to demonstrate a clear determination of preferred microhabitat in fall.

In contrast, upstream populations showed the greatest movement and emigration within the four study sections (Figures 6 and 7). Numbers of fish captured in November were substantially lower than comparable totals collected in July. Observed proportions (C/M) for LWC II and BC II, respectively, were .19 and .10 (Table 14). In addition to these relative reductions in total fish numbers, movement between one or more subsections was observed for 82 percent of the recaptured rock bass in LWC II (Figure 6). Only one marked fish was recaptured in section BC II, and this individual had not moved. The apparent stability of this single juvenile was not considered important when compared with the mass emigration suggested by population estimates (Appendix Tables 5-6).

Life stage also appeared to influence rock bass movement patterns (Figures 6 and 7). Of those individuals that moved, approximately 58 percent were adults and 42 percent were juveniles. Adults moved a modal distance of one subsection and most often in a downstream direction.

Four of the five adults which moved upstream were in adjacent subsections sharing a contiguous pool or run. The maximum net movement observed for any adult was in section LWC II. This individual was collected during the winter survey and had moved at least 169 m downstream. Juveniles moved a modal distance of one subsection and most often downstream. The greatest movement observed for any juvenile was also in LWC II and was at least 174 m downstream. Upstream move- ment observed for two individuals was between adjacent subsections sharing a contiguous run. Except for one individual, all rock bass which moved in Little Walker Creek relocated to subsections with a greater mean depth.

High winter flows and excessive water depths precluded winter electrofishing surveys above and below lower study sections (I) on both streams. Emigration upstream from section BC II was prevented by a beaver dam, and limited sampling produced no marked fish (n = 7) from deep, turbid pools immediately below this site. At the only site effectively surveyed, LWC II, no marked rock bass were among the sixteen fish captured in the 200-m run directly upstream from the study boundary. Fish ranged in length from less than 100 mm to 244 mm.

Six marked rock bass were collected in approximately 240 m of stream surveyed below LWC II. Two adults and four juveniles were captured from 4 to 89 m below subsection A and had moved from as far upstream as subsection C. Fifteen rock bass, ranging in length from 126 to 260 mm, were collected in the first 50 m electrofished. A total of 12 rock bass, only seven of which fall within this size range, was collected on the first run of the entire 250-m study section. When compared with the highest rock bass densities in any subsection of LWC II, this survey reach had five times more adult rock bass during the same period. Physical habitat in this 240-m survey reach was characterized by a mean depth of 52 cm, mean current velocity of 26 cm/sec, and predominantly cobble substrates (60 percent).

Population Dynamics

Population estimates were relatively low and variable, with adults outnumbering juveniles in nearly all subsections of both study streams (Appendix Tables 5-6). Overall variation in adult population estimates, as expressed by 95-percent confidence interval width [(CIW)/N x 100], averaged 36 percent and ranged from 0 to 256 percent in both streams. Adult numbers were generally highest in July/August, but observed maxima and minima both occurred in November. Estimated populations during this period ranged from zero in several upstream subsections of both creeks to 32 fish in the downstream subsection BC IA. Downstream subsections of both creeks also tended to support the greatest numbers year-round. Adult rock bass abundance in all but three subsections followed a pattern of high levels in July/ August, low levels in November, and a return to high levels in June. Numbers of fish in June exceeded previous August levels in several subsections of Little Walker Creek. Most of these individuals were ripe and most likely ready to spawn.

Juvenile population estimates were consistently lower than those for adults in all but five subsections. In only two of these five subsections were a majority of juveniles observed during more than one sampling period. Mean CIW was 47 percent and ranged from 0 to 237 percent for all subsections in both streams. The maximum juvenile population estimate was 27, in subsection BC IC. Zero estimates were obtained for many subsections throughout the year, particularly the upstream subsections during November and June. With few exceptions, a general pattern of decline in juvenile abundance was observed from July 1982 to June 1983 at all study sites.

Mean annual densities of rock bass were slightly higher in Back Creek than in Little Walker Creek. Mean estimates were 411 ha^{-1} and 387 ha^{-1}, respectively. Adult densities were generally much higher than those of juveniles in nearly all subsections (Appendix Tables 5-6). The maximum adult density observed was 929 ha^{-1} in subsection II E of Little Walker Creek. The highest overall densities for adults were observed in section LWC II, followed in decreasing order by BC I, BC II, and LWC I. Densities in downstream subsections were less variable than those at upstream sites throughout the year, but almost all subsections showed marked declines from July to November 1982, with densities approaching former levels by June 1983. However, seasonal variation in adult densities did not conform to this pattern in three of the downstream subsections on Back Creek and one in the lower section of Little Walker Creek.

Juvenile densities were relatively low, exceeding adult estimates in only five subsections throughout the year (Appendix Tables 5-6). This predominance of juveniles did not persist for more than one sampling period except in subsections LWC IID and BC ID. Both of these subsections had primarily juveniles in July and November. Juvenile densities were highest overall in section BC I followed by sections LWC I, LWC II, and BC II, respectively. The maximum juvenile density in any subsection was 648 ha^{-1} in BC ID during the August 1982 sample. Zero juvenile densities were recorded frequently at all four sections on both streams and increased in frequency in successive sampling periods.

Production

Mean annual production of rock bass was essentially the same in both streams. Values observed were 1.03 g m^{-2} yr^{-1} in Little Walker Creek and 1.04 g m^{-2} yr^{-1} in

Back Creek, respectively. Production was highest in the downstream section (I) of Back Creek and lowest in the upper section (II) of this same stream (Table 15).

Annual turnover ratios (P/B) were inversely related to production and averaged 0.32 across all sites. Age 0 fish were observed to contribute from 17-60 percent of the mean annual production.

The limited number of samples and highly variable population estimates hindered reliable back calculations of initial population size of the age 0 cohort using catch data. For the purposes of comparison, however, these estimates accompany those computed using stock fecundity data (See Appendix Tables 7-10). Fecundity was used in all computations of production and was based on four ripe females with a mean fecundity of 2,167 eggs (range 1,454-2,896 eggs), mean weight of 126 g (range 86-165 g), and mean length of 180 mm (range 161-198 mm). Because of the small sample size and the likely influence of spawning behavior on sex ratios observed during the collection period, sex ratios were assumed to be equal. Age at maturity for females was not determined in this study and was assumed to be three for the purposes of estimating stock fecundity (Carlander 1977).

Cohort population estimates did not decrease consistently through time, but generally increased from August

Table 15. Mean biomass, production, and annual turnover ratios of rock bass in Little Walker and Back Creeks, with the contribution of age 0 fish to annual production.

Section		Biomass (g/m^2)	Total Production (g/m^2/yr)	Turnover Ratio (P/B)	Age 0 Production (g/m^2/yr)	Age 0 % of total
LWC	I	2.60	0.83	0.32	0.36	43
	II	4.31	1.22	0.28	0.61	50
BC	I	4.78	1.27	0.27	0.21	17
	II	2.03	.81	0.40	0.49	60

1982 to June 1983 (Appendix Tables 7-10). This pattern was often coupled with a substantial decrease in November, which was most prevalent in upstream sections. Intermediate age classes tended to dominate with the exception of the 1979 year class, which was relatively weak in both streams. The greatest number of age 0 fish was captured in the lower section of Little Walker Creek, while individuals age 5 and older were sampled most frequently in the upper section of this stream.

Mean weights and growth rates varied inconsistently for cohorts with low sample sizes (Appendix Tables 7-10). Negligible population estimates were usually associated with the youngest age classes and the November sampling period. Mean weights for age 0 fish were based on as few as two individuals and were likely less than standard errors for weights measured at streamside.

In addition, the lack of this youngest age class from samples in BC II required the use of mean weights from other study sections which may have introduced some error. Low sample sizes for older individuals, particularly in November, resulted in mean weights calculated from five fish or less in all study sections. Negative growth rates, computed using mean weights, were generally related to these small sample sizes, but it was not obvious if this bias was consistent or if it resulted in significant underestimates of production. Overall, growth rates were highest for the November to June period; section BC I was the only exception.

Production was correctly ranked by Model I HSI values for three out of four study sections (Table 16). Suitability indices were very similar for the sections incorrectly ranked by this model. The "reproduction" model component also correctly ranked three of the four stream sections evaluated.

Errors in ranking were produced by differences as low as one percent in model or component suitability index values. Conversely, sites varying in production by as much as 47 percent were ranked equally by the reproduction component. In general, however, errors in ranking were most often associated with sections BC I and LWC II, or LWC I and BC II, which differed in mean annual production by four and two percent, respectively. These differences were not considered significant because of sampling error and other sources of bias.

Table 16. Production ($g/m^{-2}/yr^{-1}$) and suitability index values for two HSI models and three model components in Little Walker and Back Creeks (section ranks in parentheses).

	Stream Section			
Variable	BC I	LWC II	LWC I	BC II
Production	1.27(1)	1.22(2)	0.83(3)	0.81(4)
HSI - 1	0.96(2)	0.97(1)	0.91(3)	0.90(4)
HSI - 2	0.95(2)	0.98(1)	0.86(4)	0.87(3)
Water Quality	0.99(1)	0.98(2)	0.88(4)	0.94(3)
Reproduction	0.95(1)	0.94(2)	0.94(2)	0.90(3)
Other	0.95(2)	0.97(1)	0.89(4)	0.90(3)

DISCUSSION

Habitat Suitability Models and Assumptions

Summer Habitat Units in both streams were more positively correlated with fall biomass of rock bass than with that in any other season. As much as 68 percent of the variation in biomass was explained by this combined measure of habitat quality and quantity. The lack of significant correlations between habitat variables in June may have been the result of spawning movements. Performance of Model I, which considered all life stages, was generally better than that of Model II. Both models tended to account for more of the variation in biomass of Back Creek than in Little Walker Creek. When habitat quality alone was related to standing stocks, the strong relationship between summer habitat and fish distribution in fall was the only one maintained.

Performance of habitat suitability models can best be discussed in terms of the underlying assumptions of the methods used. Important assumptions, affecting correlations of model outputs and measures of carrying capacity, fall into three basic categories related to: 1) habitat modeling, 2) estimates of carrying capacity, 3) validation statistics relating 1 and 2. Violations involving methods in any of these categories will influence model performance and confound interpretation, particularly when multiple violations occur.

The use of habitat models to estimate species' carrying capacity is based on multiple assumptions, the most fundamental being that habitat is the factor most limiting population abundance. This basic assumption is extremely difficult to ensure or assess unless habitats are relatively undeveloped or unimpacted, and populations are unexploited. Study sites on both streams generally met these criteria with the exception of fishing observed at lower sections in both creeks. The potential effect of "exploitation" by natural predators and interspecific competition was not evaluated. Chronic turbidity in Back Creek due to agricultural land uses was considered part of the "natural" environment regarding this initial assumption.

A second major assumption related to model applicability dictates that evaluations be conducted at the time when physical habitat is most limiting. This was assumed, on the basis of model application guidelines (Terrell et al. 1982) and draft suitability curves for rock bass (USFWS 1980b), to be the summer low flow period. Hydrographs for Walker Creek (HISARS 1983), located in an adjacent watershed, indicated that minimum discharges for the general area usually occur in August. Although study streams were not gauged, monthly observations confirmed this trend.

However, fish distribution in both study streams was most limited in the November sample and suggested this to be the most critical of the three sampling periods for this particular evaluation year. Mass emigration of fish from shallower subsections and relative stability within or movement to deeper areas, all verified by marked individuals and abundance estimates, support this conclusion. Further evidence is provided by correlation analysis of HU's and biomass which yielded the highest r^2 values for November data, 0.68 and 0.51 in Back and Little Walker Creeks, respectively. Schlosser (1982) has reported similar population dynamics in a headwater stream in Illinois. My results suggest that winter habitat can be critical for rock bass in some streams of this region, but other winter observations for this species are not available for comparison. Correlations between standing stocks and weighted usable area of four warmwater species in Oklahoma varied from insignificance in summer to $r^2 = 0.86$ ($P < 0.001$) in winter (Orth and Maughan 1982). Trial et al. (in press) concluded after habitat evaluations for several coldwater species in Maine that tests of HSI models may be significantly affected by season because habitat and carrying capacity change, and populations move. Since model validation efforts can obviously be influenced by seasonal changes in carrying capacity, and the limiting season may vary unpredictably, static evaluations during seasons incorrectly assumed to be critical (e.g. summer) may continue to frustrate model validation attempts. This practice might be avoided if each study section (i.e. subsequent x,y point in validation correlations) incorporates discrete fish populations and the habitat of their respective home ranges. In this way, the seasonally dynamic nature of carrying capacity and related fish movements would be internalized by field measurements, and thus minimize invalidating variation in resultant analyses.

Habitat models that were tested also assumed that substrate, current velocity, temperature, turbidity, and gradient were the most important physical variables affecting rock bass distribution. Two factors are important in evaluating variables included in this assumption; the ecological importance of each variable, and the efficacy of methods used to measure them.

Gradient and turbidity measures were rated as optimal in virtually all study subsections of both streams and therefore were relatively unimportant in explaining observed variation in rock bass distribution. SI curves for both of these parameters (Appendix Figure 1) indicate wide optimal ranges with little sensitivity to all values measured in this study. Thus, it is unlikely that methods of measurement used for either variable had a significant impact on realistic ratings of associated habitat. Instead, it is more probable that these two variables do not change sufficiently along study reaches to have a measurable influence on the microhabitat distribution of rock bass or other fish species. This conclusion is consistent with longitudinal zonation principles described by Hynes (1970) for lotic ecosystems.

Temperature variables also received optimal or near optimal ratings in all subsections and were rarely limiting in overall estimates of habitat suitability. The notable exceptions to this tread were for three subsections at LWC I where maximum stream temperatures in summer exceeded the optimal range for this species and presumably imposed a water quality constraint on habitat quality. These high temperatures probably reflect the increased surface area and reduced canopy cover in this study reach and support to some degree the ecological accuracy of this variable. However, the fact that maximum and most suitable temperatures were of little widespread importance in delineating rock bass distribution is likely due to their observed homogeneity at all levels of comparison. Temperature ranges and maxima were similar at all four sections on both streams. Thus their low variability, and subsequent lack of utility in tested models, is more likely a function of natural homogeneity at the stream and section levels of analysis than an artifact of sampling methods and assumptions. However, measuring temperature in more thermally stable pools would not explain greater, potentially sub-optimal ranges expected in shallower subsections of a study reach. Without adjustments using pool-to-riffle ratios, or some similar geomorphic parameter, temperature suitability indices would probably be inflated in subsections dominated by shallower, riffle habitat for which "pool" temperatures were assumed. Preferred temperature experiments with rock bass (Cherry et al. 1977) indicate that this species will respond to changes of 3° C from acclimation temperatures. This behavioral response combined with known longitudinal and diurnal variation patterns in stream temperature, as much as 6° C in small streams in summer (Hynes 1970), suggest that temperature-related movements would likely occur within shallower study subsections. Such inherent variability in water temperature and the related fish movements would invariably confound evaluations of habitat suitability based on estimates of standing stocks within relatively small subsections. The assumption that optimal spawning temperatures prevailed in all subsections during the reproductive period is also questionable when stochastic variation in temperature profiles is considered. Because early life stages of fish are more sensitive to environmental changes (Snyder 1983), errors in this latter assumption could have resulted in substantial positive biases in assigned suitability indices for some subsections. Consequently, if the importance of temperature to habitat suitability is to be accurately modeled and assessed, individual study units (eg. subsections) should encompass the range of temperature-induced movements within each unit, and a range of suboptimal to optimal temperatures among all units. Based on seasonal movement data from this study, such a unit would have included a complete riffle-run-pool sequence and would have been approximately forty-five times the width of the channel. This is much longer than the repeating riffle-pool sequence of Leopold et al. (1964) which averages five to seven times the channel width. Several streams over a wide geographic area would probably be needed to provide a complete range of unsuitable to optimal temperatures between all such units.

Current velocity was the variable most frequently limiting habitat suitability for rock bass, especially juvenile life stages. Swifter currents were usually the cause of reduced suitability for both juveniles and adults, the former being more limited. Minimum current velocities in each subsection at depths of 10 cm or more were almost always zero and rated as optimal for embryo and larval life stages. This uniformly optimal rating is ecologically untenable and resulted from the cellular sampling method used. With the relatively high number of cells in each subsection, the probability of measuring a zero velocity (e.g. behind a boulder or other obstruction) was high and was ultimately responsible for the positive bias in current suitability indices for these earliest life stages. Mean current velocities associated with the highest densities of rock bass in this study ranged from 3.2 to 11.6 cm/sec, with juveniles utilizing slightly slower currents. Overall, measured current velocity probably characterizes adult and juvenile habitat accurately, but interpretation of true suitability is limited by the lack of comparable, quantitative data on this parameter. General references to preferred current velocities of rock bass are limited to "permanent flows of slight to moderate current" (Scott and Crossman 1973; Pflieger 1975; Lee et al. 1980). Hallam (1959) found that in Ontario streams of more than 10 cfs, rock bass inhabited reaches comprised of approximately 46% "still water". In a recent study on habitat use by centrarchids, Probst (1983) reported no significant differences by size class or time of day in current velocities used by rock bass. Current velocities utilized by this species averaged 9.6 cm/sec and were significantly different than those used by smallmouth bass (*Micropterus dolomieui*) and longear sunfish (*Lepomis megalotis*). Such data generally conform to draft suitability curves, but collectively suggest that interspersion of areas with slight to moderate currents might also be useful in determinations of habitat suitability for rock bass in streams. Thus, a suitability curve based on a quantitative ratio of riffle-to-pool current velocities might provide a satisfactory method for evaluating the suitability of a current interspersion factor on populations of stream-dwelling rock bass.

Spawning substrate was the second most limiting variable in habitat suitability models. Reduced suitability of this variable in LWC II and BC I resulted from a preponderance of suboptimally coarse and fine sediments, respectively. Tested models did not include substrate variables for other life stages, which may have biased estimates of habitat suitability if this parameter was limiting. The cellular sampling design alone probably did not prevent accurate classification of dominant substrates, but may have obscured some potentially important ecological relationships (e.g. combined suitability of spawning substrates and current velocities within a particular cell). Overall suitability ratings were based on means of variables computed independently and assumed that combinations of substrate and current velocity were unimportant in determining spawning habitat suitability (i.e. "reproduction" component suitability). This assumption could not be tested reliably because substrate measures were not continuous and would have violated assumptions of associated statistical procedures. Orth and Maughan (1982) reported violations of the assumption of variable independence for depth-velocity interactions, but not for velocity-substrate relationships for three bottom-dwelling fish species. However, current velocities close to the stream bottom are invariably influenced by larger substrate types and tend to confound interpretation of substrate utilization data. This is probably true of rock bass which tend to position themselves within 10 cm of the bottom (Probst et al. 1984). Even though microhabitat relationships related to substrate are not well investigated, utilization patterns observed for rock bass have been significantly different between substrate types (Probst 1983) and support the potential utility of this parameter in habitat models for this species.

Habitat models tested in this study assumed that the variables included were the most limiting to rock bass, but utilization data from this and other studies suggest that cover and depth may be equally important. Rock bass densities were consistently highest in study subsections containing the most cover (20-29%). Underwater observations in Ozark streams indicated that 76% of all rock bass observed were located `in cover', rootwads being the most utilized cover type (Probst et al. 1984). Angermeier and Karr (1984) also observed significantly higher numbers of age 2 and older rock bass in stream reaches with woody debris compared to those with no debris. Similarly, mean depth has been related to rock bass distribution as well. Area greater than 25 cm deep was the single most important variable in multiple regression models, explaining up to 91% of the variation in rock bass standing stocks in this study. Water depth and rock bass size were positively correlated in streams investigated by Probst et al. (1984). Fausch and Parsons (1984) reported that depth and instream cover were among the three most used habitat variables in all models reviewed which predict standing stock of stream fishes. Collectively, these studies infer that quantitative data on these microhabitat variables are needed if reliable fish-habitat models are to be developed for rock bass.

As with habitat measurements and modeling, violations of assumptions related to standing stock estimates may have also influenced model validation. Though correlation statistics assume that the dependent variable is measured with error, variability in subsection population estimates was probably too great to result in significant correlation with summer habitat suitability indices. Catch-effort techniques used in this study assumed closed fish populations, equal probability of capture for all individuals, and constant fishing effort during each removal period (Moran 1951). Constant fishing effort can be reasonably assumed because of the direct control over this assumption. Thus, errors in population estimates were more likely due to violations of the first two assumptions.

The assumption that rock bass populations were closed during any particular sample is reasonable, but is clearly invalid when multiple seasons are considered. It is important to remember that the HEP approach is based on a single evaluation at a time when physical habitat is presumably limiting. However, population estimates were highly variable among seasons due to immigration and emigration, and therefore could not be considered "closed" for the purposes of this analysis. The only population estimates that conformed to this assumption were those of summer which were made concurrent with the habitat evaluation. No significant correlations between these estimates and habitat quality were observed. Possible explanations for the lack of correlation are: 1) physical habitat was not limiting (i.e. populations not at carrying capacity), 2) limiting habitat variables were not measured or modeled, or 3) estimates of carrying capacity were not accurate. All three of these explanations are supported by sampling results, but to what extent they are individually responsible is unknown.

Gear selectivity undoubtedly affected the assumption of equal probability of capture for all individuals and contributed to errors in population numbers (ie. carrying capacity). Incomplete galvanotaxis of smaller size classes was frequently observed to reduce their capture efficiency. Even when effectively immobilized, smaller individuals were often irretrievable from dense cover types or lost in leaf litter and sampling-induced turbidity associated with slower stream habitats. Only adult rock bass were fully recruited to electrofishing gear. Such selectivity due to fish size and environmental factors is well documented for this gear type (Larimore 1961; Reynolds 1983). Knowing that smaller size classes were underestimated, validation of habitat suitability for the juvenile life stages is obviously limited. Layher and Maughan (in press) recognized the influence of gear selectivity on fish sampling and observed no significant correlations between standing stock and habitat variables for eight warmwater species until corrections for gear type were made. Because variables affecting gear efficiency were not always known or measured in this study, it is impossible to determine the degree of error in population estimates for the earlier life stages. Intensive sampling of larval and juvenile life stages in conjunction with habitat evaluations may prove necessary if realistic habitat suitability models are to be developed and validated.

Ultimately, errors or excessive variability in habitat and carrying capacity measures are manifested in validation statistics and often result in poor model performance. Because regression techniques assume that the independent variable (e.g. HSI) is measured without error (Lapin 1975), unknown variability in it and the dependent variable can also lead to ambiguous interpretations. It is within this framework that overall model performance is discussed.

Relative homogeneity of habitat (independent variable) and excessive variability in fish abundance (dependent variable) resulted in few consistent correlations between measures of habitat quality and performance of rock bass. Suitability indices for model components and individual variables, particularly at the subsection level, reflected this homogeneity in habitat quality and the subsequent lack of discrimination provided by overall HSI's. In addition, several violations of habitat model assumptions combined with inherent fish sampling biases undoubtedly influenced tests of assumed fish-habitat relationships.

Correlations of summer habitat and rock bass abundance ($P<.10$) were most affected by differences between study streams. Regardless of other factors considered, the number of significant correlations between fish-habitat parameters was fifty percent higher in Back Creek. It is unclear if model biases for habitat types in this stream or violations of model assumptions are responsible. The only exception to this trend was the single correlation between HSI's and standing stocks of rock bass in June observed in Little Walker Creek only. Interpretation of this anomaly is confounded by the earlier spawning movements observed in Little Walker Creek and probable sampling biases related to greater water clarity in this stream at the time. The higher correlations in Back Creek are probably an artifact of greater habitat variability and statistical methods because sampling techniques were the same in both streams. Fausch and Parsons (1984) addressed this phenomenon in a review of habitat-standing stock models and noted that wider spacing of the independent variable tends to inflate coefficients of determination (r^2). Although the range of the independent habitat variable was somewhat narrower for Back Creek, a much higher range observed in the dependent variable, population densities or standing stocks in this case, would produce the same result in computations of r^2 according to standard formulae (Lapin 1975). Similarly, multiplying HSI's and standing stocks by area (i.e. to compute HU's) also increased variability of this parameter by subsection and probably influenced resultant correlations. The obvious sensitivity of correlative relationships to variations in measurements of the independent variable suggest that unknown errors will seriously confound satisfactory interpretation or validation of habitat models using current assumptions.

Seasonal variation in rock bass abundance, the dependent variable, accounted for the greatest change in the number of significant correlations between population and habitat parameters. The strongest correlations were observed for population estimates in November. Summer

HU's explained as much as 68 percent of the rock bass biomass in Back Creek during this month, which was also the only time that HU's and biomass in Little Walker Creek were significantly correlated ($r^2 = 0.51$). These results suggest that measured habitat in summer was most limiting for rock bass populations in both study streams in November. Fall movement and abundance data further support that fish distribution was most limited during this sampling period. Similar fish-habit dynamics have been reported by Schlosser (1982) for a warm, headwater stream in Illinois. Thus, the lack of correlation between fish-habitat parameters in June and August may have resulted because most habitat was suitable during those months. It may be that winter ice in shallow, upstream areas was more limiting than maximum water temperatures in summer in these same reaches, but no winter habitat data for this species were available. Home range and seasonal movements of rock bass have been evaluated by other researchers and have varied from single pools of 100-200 feet during the summer (Gerking 1953) to distances of over 3.5 miles over four years in a large river (Brown 1960). However, rock bass have been observed to be relatively sedentary (Scott 1949; Gerking 1950, 1953; Funk 1957; Brown 1960) and thus reasonable attempts to incorporate their annual home range in habitat evaluation reaches should reduce movement- related variability in population estimates and ultimately improve model correlations.

Production and Macrohabitat

Negatively biased population estimates, particularly for younger age classes, probably resulted in substantial underestimates of mean annual production in both streams. Observed sampling biases and resultant population data indicate that the youngest age classes were not fully recruited to the electrofishing gear and thus their total biomass would have been underestimated in computations of production. In a review of fish production in warmwater streams, Neves (1981) suggested that such errors can be significant because the age 0 cohort alone may contribute as much as 84% of the total annual production for some species. Adult biomass estimates, and thus total production, were also probably low due to decreased sampling efficiency in the deeper subsections and the negative bias of the Carle and Strub (1978) population estimates as apparent from its skewed confidence intervals. Minimal sample sizes during the fall surveys also resulted in highly variable and often negative growth rates which cast further suspicion on overall production estimates.

The small sample size in this study and the scarcity of comparable production data for this species make interpretation somewhat speculative. Mean annual production of rock bass in this study was 1.04 g m^{-2} yr^{-1}, considerably higher than the 0.24 g m^{-2} yr^{-1} reported for this species in the upper Speed River, Ontario (Mahon et al.

1979). Estimates in this latter study, however, were based on only three individuals older than age 0 and confound the potential influence that more northern latitudes may have on this species. Recent production estimates for rock bass in two Missouri rivers averaged 1.40 g m^{-2} yr^{-1} (Covington et al. 1983) and were slightly higher than those observed in this study. These higher estimates included only fish age 3 and older and were determined using the method of Allen (1951). Both of these procedural differences complicate more comprehensive comparison, but do suggest that the Missouri estimates are probably minimal without data for the earliest age classes.

Turnover ratios (P/B) were relatively low (mean=0.32) for rock bass in this study and probably reflect the slower growth rates of older rock bass predominating samples from both study streams. All study sections had six or more cohorts, a condition which Chapman (1978) suggested may commonly result in P/B ratios less than one. Similarly, Waters (1977) reported a mean annual P/B ratio of 0.6 for relatively long-lived warmwater fish ($\leq$ 10 years) while a higher mean P/B ratio of 1.2 was more typical of salmonids with shorter life spans ($\leq$ 3 years). Neves (1981) discussed this inverse correlation between P/B ratios and cohort age, and further noted that carnivores generally have lower P/B ratios associated with their larger size and longevity. These findings tend to support the accuracy of the relatively low P/B ratios observed for rock bass in this study. Consistent sampling biases for larger, slower-growing individuals may have ostensibly reduced P/B ratios as well.

When HSI values and annual production data were compared, model I HSI's correctly ranked production at three of the four sections in this study. Differences in habitat measures were so slight that such ranking would unlikely remain significant after standard statistical testing. This reasonably accurate ranking seems even more fortuitous when the cumulative assumptions of all techniques are considered. Farmer et al. (1982) cautioned that acceptable relationships between habitat model outputs and animal abundance become less likely as the number of assumptions increases. The potential and observed violations of assumptions associated with both of these parameters in this study suggest that such "whole model" validations will be suspect until individual assumptions are verified independently over a wide range of conditions.

In general, minimum depth parameters were strongly correlated with rock bass distribution in this study and may be an important addition to future habitat models for this stream-dwelling species in southern portions of its range. Comparatively high annual production in the two Missouri rivers (Covington et al. 1983) also suggests that depth and cover parameters be added to HSI models for this species if habitat-related productivity is to be fully considered. However, until a modicum of quantitative habitat and production data for rock bass become available evaluation of fish-habitat models for this species will be only approximations.

SUMMARY

1) Significant rank correlations ($P < .10$) between biomass and Habitat Units were observed at all sites in fall and at no sites during spring.
2) Correlations between biomass and HU's were consistently highest in Back Creek using model I and biomass estimates in November.
3) Current velocity (juvenile) and substrate (embryo) were ultimately the most important variables in determining relative habitat suitability using HSI models.
4) Substantially reduced standing stocks, resulting from the downstream emigration of many individuals to deeper pools, were observed at both upstream sites in fall.
5) The best fitting [R^2, $C(P)$] and most significant (P-value) multiple regression models were produced using fall standing stocks and included the single depth variable X6 (area > 25 cm in depth).
6) Annual production was essentially the same in both streams and averaged 1.04 g m^{-2} yr^{-1}.

RECOMMENDATIONS

1) Field validation studies of HSI models must first attempt to meet the principal assumption that it is the habitat which is actually limiting species performance (e.g. standing stocks). Fishing, spawning activity, fluctuating physical parameters, and the sampling methods themselves will all influence measured indices of carrying capacity and will likely confound model validation efforts if not considered.

2) Researchers must also ensure that habitat is limiting species performance at the time of the evaluation. It is often assumed that summer low flows are most limiting in lotic environments when in fact winter may be the critical period in some years. Temporal and geographic variability of the "limiting" period for a species needs to be incorporated or considered in model development and validation research.

3) For evaluation purposes, the selection of ecologically significant and representative stream reaches (i.e. a reach containing the approximate "home range" of the species or population being sampled), as opposed to the often smaller sections chosen to reduce costs and expedite sampling, would help mitigate the confounding influences of diurnal, environmentally-induced, or

seasonal fish movements and thus yield more meaningful results pertaining to the actual suitability of the habitat.

4) Relative habitat homogeneity should be ensured within each subsection assigned a single suitability index value. Otherwise, the unexpected presence of the species or high standing stocks may not be accounted for in an area which received a low suitability index value, but contained a favorable "refugium" or preferred habitat type.

5) The development of a variance estimate for all model outputs (e.g. Habitat Units) would facilitate the interpretation and evaluation of subsequent validation efforts.

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APPENDIX

Appendix Table 1. Physical habitat measurements and suitability index (SI) values for study section I on Little Walker Creek.

Variable		Subsection A data	A SI	B data	B SI	C data	C SI	D data	D SI	E data	E SI
turbidity (FTU)	(V_1)	3.6	1.00	3.6	1.00	3.6	1.00	3.6	1.00	3.6	1.00
	(V_2)	2.5	1.00	2.5	1.00	2.5	1.00	2.5	1.00	2.5	1.00
	(V_3)	5.0	1.00	5.0	1.00	5.0	1.00	5.0	1.00	5.0	1.00
	(V_4)	3.6	1.00	3.6	1.00	3.6	1.00	3.6	1.00	3.6	1.00
temperature (^{o}C)	(V_5)	29.0	0.78	29.0	0.78	29.0	0.78	29.0	0.78	29.0	0.78
	(V_6)[a]	-	1.00	-	1.00	-	1.00	-	1.00	-	1.00
	(V_7)	29.0	0.75	29.0	0.75	29.0	0.75	29.0	0.75	29.0	0.75
	(V_8)	29.0	0.78	29.0	0.78	29.0	0.78	29.0	0.78	29.0	0.78

Appendix Table 1. (continued)

Variable		Subsection A data	A SI	B data	B SI	C data	C SI	D data	D SI	E data	E SI
current velocity (cm/sec)	(V_9)	15.8	0.78	12.5	0.85	0	0.50	0.6	0.75	1.8	1.00
	(V_{10})	2	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{11})	2	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{12})	15.8	0.30	12.5	0.40	0	0.50	0.6	0.60	1.8	1.00
substrate[b]	(V_{13})	6(4)	0.70	4(5)	0.74	4(3)	0.75	4(3)	0.71	4(3)	0.70
	(V_{14})	6(4)	0.68	4(5)	0.74	4(3)	0.86	4(3)	0.86	4(3)	0.86
	(V_{15})	6(4)	0.49	4(5)	0.68	4(3)	0.79	4(3)	0.76	4(3)	0.75
	(V_{16})	6(4)	0.69	4(5)	0.70	4(3)	0.82	4(3)	0.80	4(3)	0.79
gradient (m/km)	(V_{17})	15	1.00	20	1.00	10	1.00	10	1.00	10	1.00

[a]assumed to be optimal (SI=1.00) at some point during spring/summer period

[b]secondary substrate type in parentheses

Appendix Table 2. Physical habitat measurements and suitability index (SI) values for study section II on Little Walker Creek.

Variable		Subsection A data	SI	B data	SI	C data	SI	D data	SI	E data	SI
turbidity (FTU)	(V_1)	4.5	1.00	4.5	1.00	4.5	1.00	4.5	1.00	4.5	1.00
	(V_2)	3.5	1.00	3.5	1.00	3.5	1.00	3.5	1.00	3.5	1.00
	(V_3)	5.0	1.00	5.0	1.00	5.0	1.00	5.0	1.00	5.0	1.00
	(V_4)	4.5	1.00	4.5	1.00	4.5	1.00	4.5	1.00	4.5	1.00
temperature (^{o}C)	(V_5)	27.0	1.00	27.0	1.00	27.0	1.00	27.0	1.00	27.0	1.00
	(V_6)[a]	-	1.00	-	1.00	-	1.00	-	1.00	-	1.00
	(V_7)	27.0	0.90	27.0	0.90	27.0	0.90	27.0	0.90	27.0	0.90
	(V_8)	27.0	1.00	27.0	1.00	27.0	1.00	27.0	1.00	27.0	1.00

Appendix Table 2. (continued)

Variable		Subsection									
		A		B		C		D		E	
		data	SI	data	SI	data	SI	data	SI	data	SI
current velocity (cm/sec)	(V_9)	3.7	1.00	8.8	1.00	8.9	1.00	5.4	1.00	3.9	1.00
	(V_{10})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{11})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{12})	3.7	1.00	8.8	0.60	8.9	0.60	5.4	1.00	3.9	1.00
substrate[b]	(V_{13})	4(5)	0.78	5(4)	0.90	5(4)	0.90	4(5)	0.80	4(5)	0.79
	(V_{14})	4(5)	0.83	5(4)	0.75	5(4)	0.75	4(5)	0.82	4(5)	0.82
	(V_{15})	4(5)	0.74	5(4)	0.70	5(4)	0.63	4(5)	0.75	4(5)	0.74
	(V_{16})	4(5)	0.86	5(4)	0.78	5(4)	0.80	4(5)	0.81	4(5)	0.77
gradient (m/km)	(V_{17})	15	1.00	15	1.00	15	1.00	10	1.00	10	1.00

[a]assumed to be optimal (SI=1.00) at some point during spring/summer period

[b]secondary substrate type in parentheses

Appendix Table 3. Physical habitat measurements and suitability index (SI) values for study section I on Back Creek.

		Subsection									
		A		B		C		D		E	
Variable		data	SI	data	SI	data	SI	data	SI	data	SI
turbidity (FTU)	(V_1)	10.9	1.00	10.9	1.00	10.9	1.00	10.9	1.00	10.9	1.00
	(V_2)	14.0	1.00	14.0	1.00	14.0	1.00	14.0	1.00	14.0	1.00
	(V_3)	8.0	1.00	8.0	1.00	8.0	1.00	8.0	1.00	8.0	1.00
	(V_4)	10.9	1.00	10.9	1.00	10.9	1.00	10.9	1.00	10.9	1.00
temperature (°C)	(V_5)	26.0	0.96	26.0	0.96	26.0	0.96	26.0	0.96	26.0	0.96
	(V_6)[a]	-	1.00	-	1.00	-	1.00	-	1.00	-	1.00
	(V_7)	26.0	1.00	26.0	1.00	26.0	1.00	26.0	1.00	26.0	1.00
	(V_8)	26.0	0.96	26.0	0.96	26.0	0.96	26.0	0.96	26.0	0.96

Appendix Table 3. (continued)

Variable		Subsection A data	A SI	Subsection B data	B SI	Subsection C data	C SI	Subsection D data	D SI	Subsection E data	E SI
current velocity (cm/sec)	(V_9)	1.6	1.00	4.3	1.00	5.4	1.00	19.1	0.70	11.4	0.94
	(V_{10})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{11})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{12})	1.6	1.00	4.3	1.00	5.4	1.00	18.1	0.24	11.4	0.51
substrate[b]	(V_{13})	1(-)	0.20	1(-)	0.20	1(-)	0.20	4(1)	0.60	1(4)	0.44
	(V_{14})	1(-)	0.80	1(-)	0.80	1(-)	0.80	4(1)	0.83	1(4)	0.82
	(V_{15})	1(-)	0.50	1(-)	0.50	1(-)	0.50	4(1)	0.68	1(4)	0.61
	(V_{16})	1(-)	0.20	1(-)	0.20	1(-)	0.20	4(1)	0.64	1(4)	0.46
gradient (m/km)	(V_{17})	10	1.00	10	1.00	10	1.00	15	1.00	10	1.00

[a]assumed to be optimal (SI=1.00) at some point during spring/summer period

[b]secondary substrate type in parentheses; dash indicates no secondary substrate type observed

Appendix Table 4. Physical habitat measurements and suitability index (SI) values for study section II on Back Creek.

Variable		Subsection A data	A SI	B data	B SI	C data	C SI	D data	D SI	E data	E SI
turbidity (FTU)	(V_1)	14.1	1.00	14.1	1.00	14.1	1.00	14.1	1.00	14.1	1.00
	(V_2)	18.5	0.91	18.5	0.91	18.5	0.91	18.5	0.91	18.5	0.91
	(V_3)	22.0	0.81	22.0	0.81	22.0	0.81	22.0	0.81	22.0	0.81
	(V_4)	14.1	1.00	14.1	1.00	14.1	1.00	14.1	1.00	14.1	1.00
temperature (^{o}C)	(V_5)	25.0	0.92	25.0	0.92	25.0	0.92	25.0	0.92	25.0	0.92
	(V_6)[a]	-	1.00	-	1.00	-	1.00	-	1.00	-	1.00
	(V_7)	25.0	1.00	25.0	1.00	25.0	1.00	25.0	1.00	26.0	1.00
	(V_8)	25.0	0.92	25.0	0.92	25.0	0.92	25.0	0.92	25.0	0.92

Appendix Table 4. (continued)

Variable		A data	A SI	B data	B SI	C data	C SI	D data	D SI	E data	E SI
		Subsection									
current velocity (cm/sec)	(V_9)	9.2	1.00	24.4	0.56	14.4	0.82	13.3	0.86	18.6	0.69
	(V_{10})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{11})	0	1.00	0	1.00	0	1.00	0	1.00	0	1.00
	(V_{12})	9.2	0.60	24.4	0.10	14.4	0.38	13.3	0.40	18.6	0.21
substrate[b]	(V_{13})	1(3)	0.42	4(3)	0.66	1(6)	0.46	6(1)	0.48	6(1)	0.50
	(V_{14})	1(3)	0.88	4(3)	0.90	1(6)	0.69	6(1)	0.68	6(1)	0.55
	(V_{15})	1(3)	0.69	4(3)	0.79	1(6)	0.55	6(1)	0.56	6(1)	0.50
	(V_{16})	1(3)	0.47	4(3)	0.74	1(6)	0.45	6(1)	0.50	6(1)	0.45
gradient (m/km)	(V_{17})	10	1.00	15	1.00	10	1.00	5	1.00	15	1.00

[a]assumed to be optimal (SI=1.00) at some point during spring/summer period

[b]secondary substrate type in parentheses

Table 5. Population estimates (N), percent confidence interval width (CIW expressed as a percent of N), and densities (N/ha) of rock bass in Little Walker Creek during 1982-1983.

Section	Subsection	Size class	July 1982			November 1982			June 1983		
			N	(CIW)	N/ha	N	(CIW)	N/ha	N	(CIW)	N/ha
I	A	A	9	(11)	350	4	(100)	156	7	(29)	272
		J	0	(-)	0	1	(0)	39	1	(0)	39
	B	A	28	(50)	753	7	(14)	188	20	(10)	538
		J	7	(29)	188	1	(0)	27	3	(0)	81
	C	A	12	(17)	214	11	(18)	196	18	(11)	321
		J	7	(29)	125	4	(50)	71	1	(100)	18
	D	A	14	(71)	186	21	(38)	278	28	(111)	371
		J	19	(237)	252	8	(25)	106	3	(33)	40
	E	A	12	(50)	144	6	(33)	72	9	(22)	108
		J	8	(100)	96	11	(18)	132	0	(-)	0
	Mean	A			329			178			322
		J			132			75			36
II	A	A	21	(100)	461	5	(0)	110	23	(9)	505
		J	0	(-)	0	0	(-)	0	2	(100)	44
	B	A	10	(10)	297	0	(-)	0	15	(7)	446
		J	0	(-)	0	0	(-)	0	0	(-)	0
	C	A	19	(47)	746	0	(-)	0	12	(50)	471
		J	1	(100)	39	0	(-)	0	0	(-)	0
	D	A	4	(25)	118	1	(0)	29	10	(20)	295
		J	6	(150)	177	4	(50)	118	4	(25)	118
	E	A	24	(8)	929	6	(0)	232	22	(23)	851
		J	7	(29)	271	0	(-)	0	0	(-)	0
	Mean	A			510			74			514
		J			97			24			32

Table 6. Population estimates (N), percent confidence interval width (CIW expressed as a percent of N), and densities (N/ha) of rock bass in Back Creek during 1982-1983.

Section	Subsection	Size class	July 1982 N	July 1982 (CIW)	July 1982 N/ha	November 1982 N	November 1982 (CIW)	November 1982 N/ha	June 1983 N	June 1983 (CIW)	June 1983 N/ha
I	A	A	12	(92)	214	32	(31)	571	6	(50)	107
		J	15	(33)	268	7	(29)	125	2	(0)	36
	B	A	27	(67)	503	31	(16)	577	4	(0)	75
		J	8	(13)	149	10	(20)	186	0	(-)	0
	C	A	30	(33)	558	18	(28)	335	18	(50)	335
		J	27	(96)	502	8	(100)	149	1	(0)	19
	D	A	20	(10)	518	3	(0)	78	16	(31)	415
		J	25	(32)	648	8	(188)	207	0	(-)	0
	E	A	21	(76)	504	4	(75)	96	14	(36)	336
		J	16	(13)	384	0	(-)	0	1	(0)	24
	Mean	A			459			331			254
		J			390			133			16
II	A	A	9	(256)	248	0	(-)	0	3	(33)	83
		J	0	(-)	0	0	(-)	0	0	(-)	0
	B	A	8	(25)	360	0	(-)	0	11	(18)	495
		J	2	(50)	90	0	(-)	0	0	(-)	0
	C	A	7	(14)	247	1	(0)	35	6	(17)	211
		J	2	(0)	70	0	(-)	0	0	(-)	0
	D	A	14	(14)	652	2	(50)	93	17	(29)	792
		J	1	(0)	47	0	(-)	0	0	(-)	0
	E	A	10	(20)	443	2	(50)	89	10	(0)	443
		J	0	(-)	0	0	(-)	0	0	(-)	0
	Mean	A			390			43			405
		J			41			0			0

Appendix Table 7. Calculations of production of rock bass in section I of Little Walker Creek

Cohort	Time	W(g)	N	G	B(g)	P(g)
1982	Emergence	0.008	67,177 (81)[b]			
				3.44	268.71	924.36
	August	0.25[a]	0 (76)			
				0.73	2.34	1.71
	November	0.52	9 (61)			
				1.38	55.00	75.90
	June	2.06	8 (38)			
						1,001.97
1981	August	3.39	41			
				0.63	120.54	75.94
	November	6.38	16			
				0.56	163.24	91.41
	June	11.22	20			
						167.35
1980	August	14.05	34			
				0.37	340.15	125.86
	November	20.26	10			
				0.66	648.28	427.86
	June	39.07	28			
						553.72
1979	August	36.93	7			
				0.18	306.62	55.19
	November	44.43	8			
				0.25	746.96	186.74
	June	56.96	20			
						241.93
1978	August	69.64	29			
				-0.01	1,770.65	-17.71
	November	69.17	22			
				0.33	1,388.58	458.23
	June	96.57	13			
						440.52
1977	August	148.55	5			
				-0.34	844.55	-287.15
	November	105.15	9			
				0.33	546.02	180.19
	June	145.70	1			
						-106.96
				TOTAL POPULATION		2,298.53

[a] Estimated from LWCII

[b] Back-calculated population estimates assuming constant survival of the 1981 cohort.

Appendix Table 8. Calculations of production of rock bass in section II of Little Walker Creek

Cohort	Time	W(g)	N	G	B(g)	P(g)
1982	Emergence	0.008	71,511 (32)[b]			
				3.44	286.29	984.84
	August	0.25	2 (29)			
				1.45	1.86	2.70
	November	1.07	3 (24)			
				1.05	10.82	11.36
	June	3.07	6 (15)			
						998.90
1981	August	4.43	12			
				-0.03	28.73	-0.86
	November	4.30	1			
				1.13	115.63	130.66
	June	13.35	17			
						129.80
1980	August	14.95	20			
				0.08	181.86	14.55
	November	16.18	4			
				0.67	460.85	308.77
	June	31.74	27			
						323.32
1979	August	31.80	3			
				0.27	109.91	29.68
	November	41.47	3			
				0.92	269.41	247.86
	June	103.60	4			
						277.54
1978	August	65.19	32			
	November	-	0	0.25	1714.48	428.62
	June	83.93	16			428.62
1977	August	162.00	23			
				-0.28	2170.18	-607.65
	November	122.87	5			
				0.26	1741.69	452.84
	June	159.39	18			
						-154.81
				TOTAL POPULATION		2,003.37

[b] Back-calculated population estimates assuming constant survival of the 1981 cohort.

Appendix Table 9. Calculations of production of rock bass in section I of Back Creek

Cohort	Time	$\bar{W}$(g)	N	G	$\bar{B}$(g)	P(g)
1982	Emergence	0.008	34,672 (3250)[b]			
				3.44	138.69	477.09
	August	0.25[a]	0 (2930)			
				2.52	4.65	11.72
	November	3.10	3 (2040)			
				0.72	17.39	12.53
	June	6.37	4 (99)			
						501.33
1981	August	7.98	91			
				1.05	704.04	739.24
	November	22.73	30			
				-0.17	590.16	100.33
	June	19.17	26			
						638.91
1980	August	34.31	75			
				0.56	3,468.23	1,942.21
	November	59.77	73			
				-0.18	2,807.23	-505.30
	June	50.50	25			
						1,436.91
1979	August	67.64	24			
				0.36	1,152.13	414.77
	November	97.27	7			
				-0.03	622.45	-18.67
	June	94.00	6			
						396.10
1978	August	143.32	11			
				0.12	1,431.62	171.79
	November	160.84	8			
				-0.08	717.41	-57.39
	June	148.10	1			
						114.40
				TOTAL POPULATION		3,087.65

[a] Estimated from LWCII

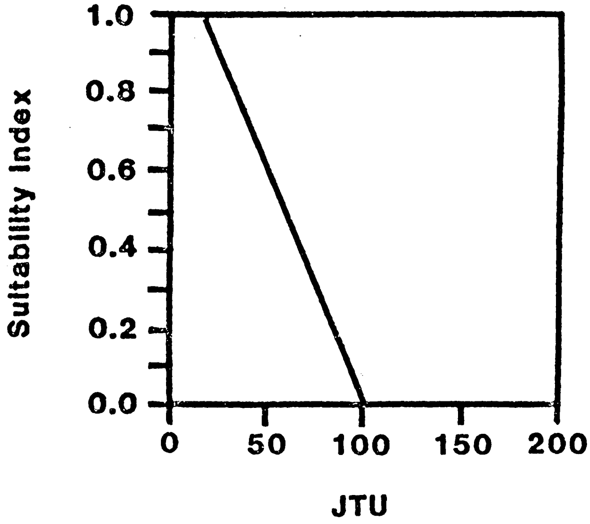
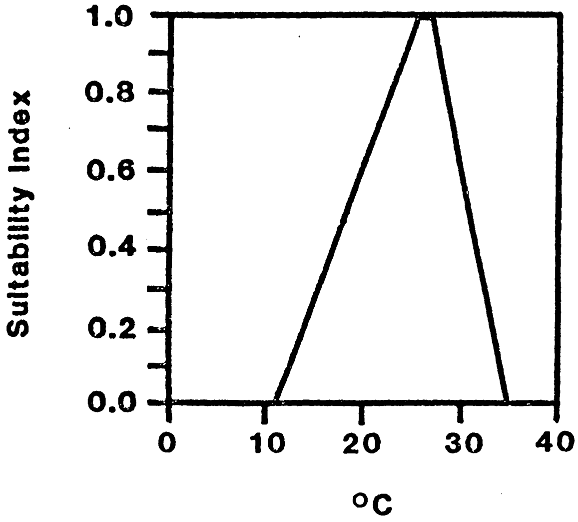
[b] Back-calculated population estimates assuming constant survival of the 1981 cohort.

Appendix Table 10. Calculations of production of rock bass in section II of Back Creek

Cohort	Time	W(g)	N	G	B(g)	P(g)
1982	Emergence	0.008	47,674 (222)[b]			
				3.44	190.70	656.01
	August	0.25[a]	0 (198)			
				2.52	0.00	0
	November	3.10	0 (137)			
				0.72	0.00	0
	June	6.37	0 (7)			
						656.01
1981	August	10.68	6			
	November	-	0	0.40	64.04	25.62
	June	16.00	4			
						25.62
1980	August	24.56	35			
				-0.46	460.80	-211.97
	November	15.50	4			
				0.83	637.56	529.17
	June	35.68	34			
						317.20
1979	August	66.70	4			
				-0.73	149.40	-109.06
	November	32.00	1			
				0.99	187.80	185.92
	June	85.90	4			
						76.86
1978	August	146.16	9			
	November	-	0	0.01	1,027.47	10.27
	June	147.90	5			
						10.27
				TOTAL POPULATION		1,085.96

[a] Estimated from LWCII

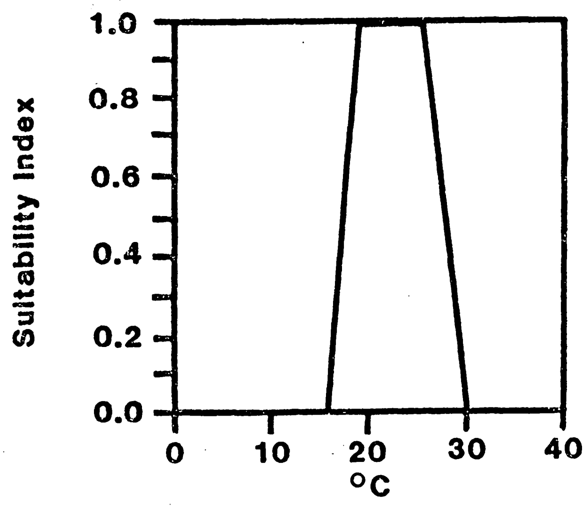
[b] Estimated from BCI

Variable	Suitability Graph
Turbidity during periods of: V_1, V_4 - normal flow V_2 - spawning period, normal flow V_3 - postspawning period, normal flow	
V_5, V_8 Maximum temperature in pools in mid-summer. (Adult, Juvenile)	

Appendix Figure 1. Rock Bass suitability index (SI) graphs for select habitat variables (from USFWS 1980a).

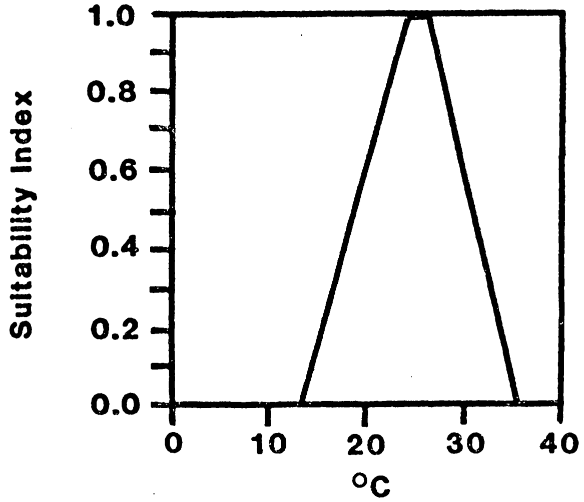
V_6

Most suitable temperature in pools during spring/summer. (Embryo)



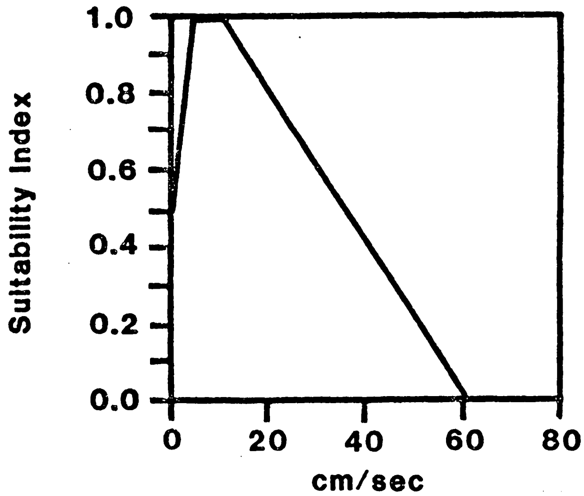
V_7

Maximum temperature in pools during the post-spawning period of early summer. (Fry)



V_9

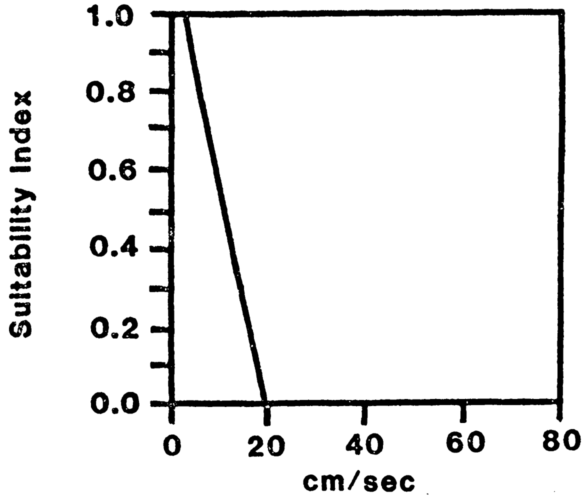
Mean current velocity in pools during periods of normal flow. (Adult)



Appendix Figure 1. (continued)

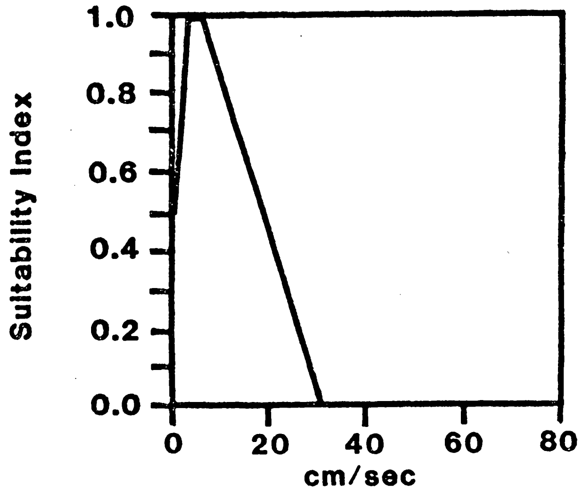
Minimum current velocity in pools
$\geq$ 10 cm deep during:
V_{10} - spawning period

V_{11} - postspawning period
(Embryo, Fry)



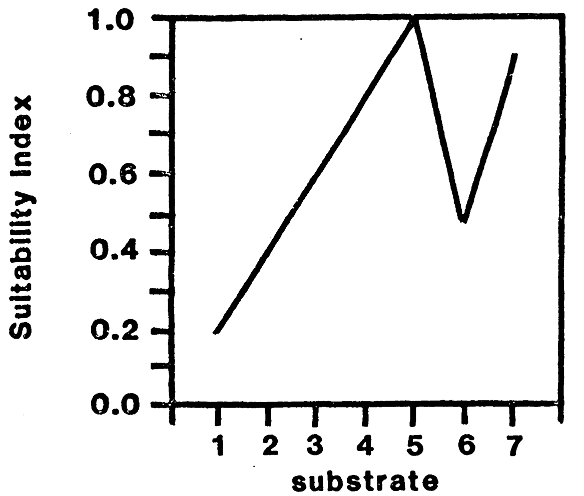
V_{12}

Mean current velocity in pools
during periods of normal flow.
(Juvenile)



V_{13}

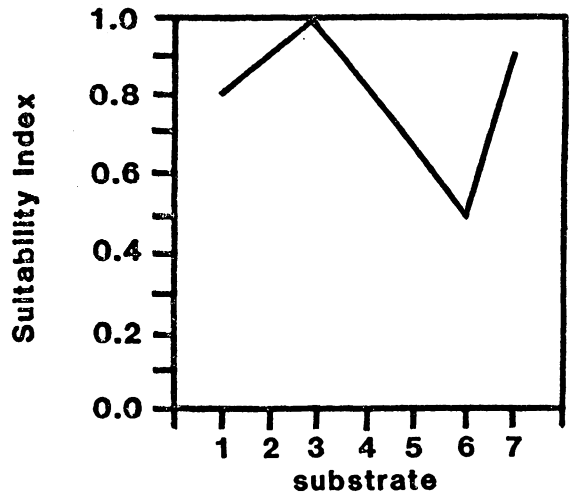
Substrate type in pools.
(Adult)



Appendix Figure 1. (continued)

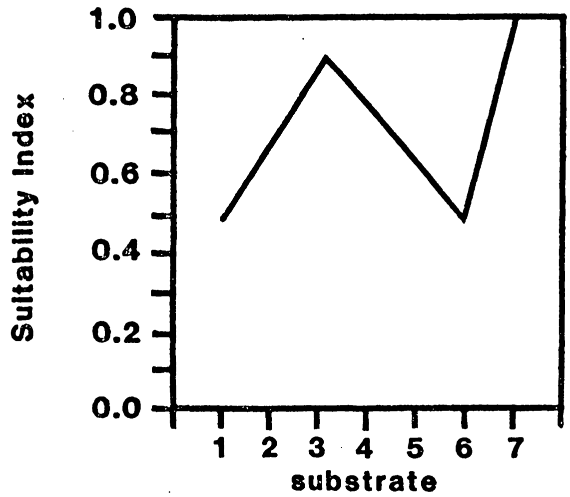
V_{14}

Substrate type in pools during spring or summer.
(Embryo)



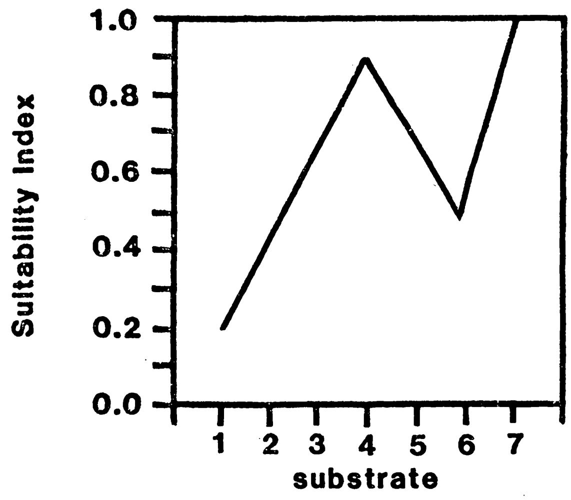
V_{15}

Substrate type in pools during spring or summer.
(Fry)



V_{16}

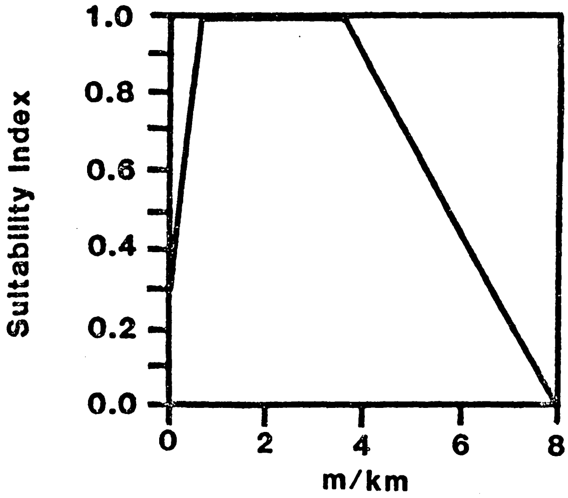
Substrate type in pools.
(Juvenile)



Appendix Figure 1. (continued)

V_{17}

Stream gradient.



Appendix Figure 1. (continued)

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